A MANUAL FOR ASSESSING RESTORED AND NATURAL COASTAL WETLANDS

WITH EXAMPLES FROM SOUTHERN CALIFORNIA

by the

Pacific Estuarine Research Laboratory

(PERL)

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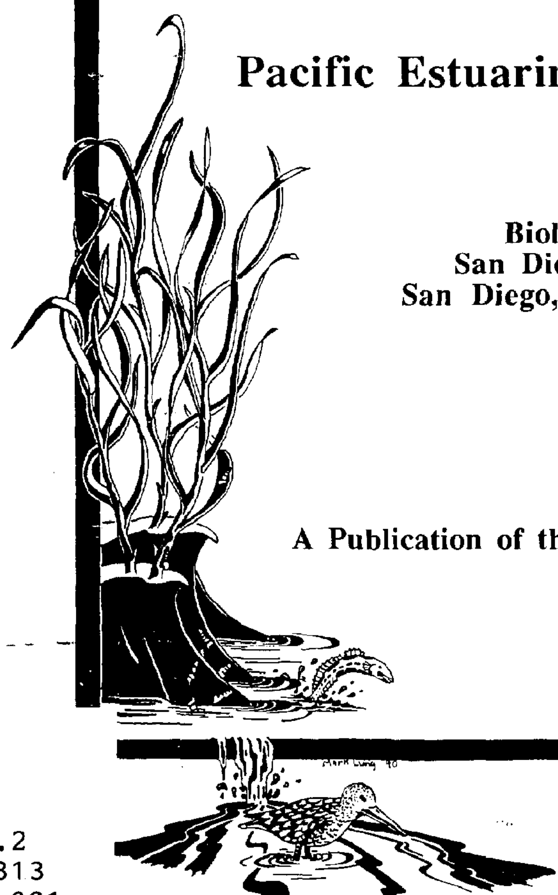
1990

A Publication of the California Sea Grant College



Dedicated to the memory of

Dr. Millicent Quammen,
estuarine researcher,
who insisted that restored wetlands
be assessed on the basis of
their functioning.

Cite as: Pacific Estuarine Research Laboratory. 1990. A manual for assessing restored and natural coastal wetlands with examples from southern California. California Sea Grant Report No. T-CSGCP-021. La Jolla, California.

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Research and Publication Support

This work is a result of research sponsored in part by the National Oceanic and Atmospheric Administration (NOAA), National Sea Grant College Program, Department of Commerce, under grant number NA85AA-D-SG140, project number R/CZ-82, through the California Sea Grant College, in part by NOAA Office of Coastal Zone Management, Marine and Estuarine Management Division, under contract no. 85-0236-85-072-A, and in part by the California State Resources Agency, the California Department of Transportation, and the California State Coastal Conservancy.

Support for publication was provided in part by the NOAA Coastal Ocean Program.

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I. Introduction

This manual presents recommendations for assessing the structure and functioning of coastal wetlands, with emphasis on salt marshes and tidal creeks. While the recommendations can be applied to many situations, the main purpose of the manual is to standardize methods of assessing restored, enhanced, or constructed wetlands.

The basic premise is that a man-made wetland should provide the same values as the region's natural wetland ecosystems. This is especially important in southern California, where native wetland habitats have dwindled to small, disturbed, isolated remnants, most of which can no longer sustain the biodiversity that once characterized this region. Our remarks often emphasize rare native species, but we do not suggest that management be restricted to target species (Zedler 1984). The threatened and endangered species that often drive restoration and management efforts are the obvious symptoms of more subtle changes in ecosystem structure and function A whole-system approach at the regional scale is needed to assess, understand, and manage coastal southern California wetlands.

With restoration and enhancement plans growing in number and size, it is imperative that we understand how well such projects are working if we are to achieve the overall objective of maintaining regional biodiversity. If most projects are failing to provide the habitat required for native species to persist in perpetuity, we must find better restoration techniques, and we must certainly halt mitigation practices that allow the "replacement" of native wetland habitat with artificially constructed wetlands.

The list of scientists whose work and advice have contributed to this manual indicates the broad range of skills and experience needed to understand complex ecosystems. In addition to drawing upon their expertise, the text includes information that has been summarized for various conferences on wetland restoration (e.g., Zedler et al. 1988, 1990), as well as information from manuscripts in progress (e.g., Nordby and Zedler, in press; Langis et al., in press) and recent studies carried out at Tijuana Estuary

The need. Thorough evaluation procedures that can stand the test of scientific review are clearly needed. Quammen (1986) recommended two types of monitoring to evaluate whether created wetlands compensate for the losses in natural wetlands: first, an assessment of compliance with resource agency recommendations and, second, long-term, scientific evaluation to test predictions and hypotheses concerning the development of natural ecosystem functions. We suggest that the two assessment goals be merged, such that "compliance" becomes the successful "replacement of lost wetland functions." Projects would be considered in compliance with mitigation policies once the constructed or modified wetland shows high potential for achieving natural functional attributes. Furthermore, the information base upon which judgments of compliance/success are made should be able to withstand scientific review. The problems resulting from erroneously concluding that a project is successful are too great for the assessment process to be casual or short-term. A margin of safety is essential.

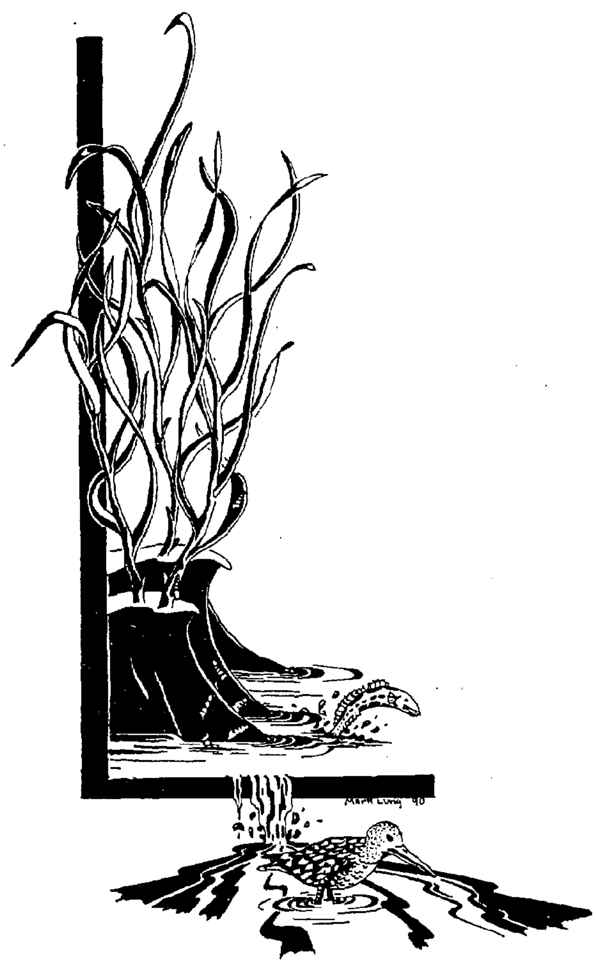
From the ecological perspective, determining when a constructed wetland has attained the functions of a natural wetland is neither simple nor quick. As Odum (1987, p. 67) points out, "Too frequently, the success or failure...is determined after a year or two's growth of the original, planted vegetation. Unfortunately, dramatic unanticipated changes may occur over the ensuing years....it is not uncommon for the plant community to become invaded and dominated by aggressive 'disturbance species'.... The long-term result may be a wetland environment which has limited functional value for wildlife habitat support or nutrient processing and which lacks aesthetic attractiveness to the degree originally planned." Likewise, Broome et al. (1987, p. 197) concluded that monitoring a constructed wetland for four growing seasons was insufficient to determine "if the created marsh reaches equivalent levels of production for all plant species and remains self-sustaining."

Organization of this manual. The rationale for requiring constructed wetlands to meet strict assessment criteria is developed in Section II. A case study of the Sweetwater River Wetlands Complex follows (Section III). For this case, functional criteria were mandated by resource agencies, and the Pacific Estuarine Research Laboratory was requested to undertake the detailed assessment. The assessment is not yet complete, but results to date show how the 4-5-year-old wetlands compare to natural wetland remnants.

The assessment methods are grouped under ten attributes of coastal wetlands (Section IV). For each component of the ecosystem, the reasons for study and assessment are provided, followed by methods recommended by PERL.

Recommendations for the minimum sampling effort needed for monitoring constructed wetlands are provided in Section V. Reference data and sources of additional information are then given.

The manual is not intended to be a final statement, but a working document that will continue to evolve as our knowledge of coastal wetlands advances. In all applications of the work, users should check with PERL to insure that the latest information is in hand.



II. Strategies for wetland construction, restoration and enhancement

1. Concerns

Coastal wetland restoration is an active endeavor in many regions of the country. New habitats are being constructed using dredge spoils, grading of topography, or other techniques. Wetland construction and restoration projects are often initiated to offset or mitigate impacts of wetland destruction. When uplands are excavated to construct habitats as replacements for lost wetlands, the question is--are the lost wetland values replaced? When existing wetlands are modified to restore or enhance wetland values, two questions arise--are existing values maintained and are additional values provided? As perceptions of success or failure have become more controversial, two things have happened--criteria for project compliance have become more detailed (as indicated by more complicated permit requirements issued by the US Army Corps of Engineers), and researchers have become more interested in studying man-made systems.

Wetland scientists around the country (cf. regional reviews in Kusler and Kentula 1989, Strickland 1986, Good 1987, Zelazney and Feierabend 1987) are concerned that restoration and habitat construction attempts are not replacing lost wetland values. Milton Weller of Texas A&M states (in Kusler and Kentula 1989), "In most situations we can provide the environmental needs to allow dominant wetland plants and animals to succeed, and the product will satisfy many if not most viewers. We cannot, however, expect to replace the complex and diverse natural systems that are a product of many centuries of evolution and randomness...."

Too many projects have emphasized single-species management. The planting of mangroves, cordgrass, or eelgrass seems to have become synonymous with tidal wetland restoration, and transplant survival rates have become measures of "success." In southern California, the endangered species law has had a curious impact on restoration activities--it has tended to suggest single-species management and to imply that one wetland type should be modified to meet the needs of a target species. Thus, we have seen proposals to plant cordgrass in pickleweed marshes, as mitigation for habitat destruction, because cordgrass provides habitat for the federally endangered light-footed clapper rail. However, pickleweed provides habitat for the State-listed Belding's Savannah sparrow, and conversion of one vegetation type to another does not constitute ecosystem enhancement.

We question the <u>hypothesis</u> that constructed wetlands are carrying out the broad range of natural wetland <u>functions</u>. Unfortunately, existing data are inadequate for testing this hypothesis. Evaluations of both natural and constructed ecosystems have emphasized structural, rather than functional, attributes. Furthermore, most of the structural data are for plants. Fewer evaluations of man-made wetlands have considered the animal components.

Three other factors add to the difficulty of comparing man-made and natural system functions. First, there may not be suitable models with which to compare constructed systems--most coastal wetlands are already modified; their hydrologic regimes (inundation and salinity patterns) have probably changed dramatically in the past century, and key species may already have been eliminated. Is it sufficient to strive for conditions that now exist in wetland remnants, or must we understand what conditions allowed those wetland ecosystems to develop? The latter seems to be necessary for plant communities where population establishment limits biodiversity. If we lack the necessary blueprints for wetland development, we can't expect to construct a system that mimics nature.

Second, the functions of natural models have not been thoroughly studied. We have far more information on the composition of benthic macroinvertebrate communities than on their use by shorebirds and other predators. There is considerable information on nutrient levels in salt marsh soils, but less understanding of the sources and rates of inflow and outflow. Studies of one important process, i.e. the net flux of detritus, suggest that results differ from place to place. Both the direction of net movement and the magnitude of organic matter flux vary considerably from wetland to wetland, and even from site to site within a given system.

Finally, there hasn't been sufficient time for constructed wetland ecosystems to mature--hence, it is difficult to assess their future potential for performing natural wetland functions. Few man-made systems have been in place and monitored for more than 5 years. While it may be possible to construct tidal wetlands that eventually mimic the functions of natural wetlands, judgments are often made after only 1-3 years. Because we don't know how long it takes, long-term assessments must be planned and continued until we are reasonably confident that expectations have been met. An additional reason for long-term assessment is that single measurements (one-time or one-year data sets) are unlikely to describe salt marsh structure and functioning adequately, because of high temporal variability in climate and wetland responses. Low plant biomass in one year may indicate poor potential for salt marsh development, or it may simply reflect a year of below-average growing conditions.

Many salt marsh mitigation efforts have problems. In some cases, failures are due to poor planning; in others, plans are not properly implemented. Sometimes, mitigation is required but never implemented. A recent evaluation of 11 freshwater wetlands constructed in Oregon included data on hydrology (using the presence of water and saturated soil as indicators), topography (slope), and plant species occurrences (Gwin and Kentula 1990). The vegetation at the constructed wetlands differed greatly from species lists given in the permit file. Volunteer species (those not on the planting list) made up 93-100% of the species present (ibid.). Similar results were found in evaluations of Florida wetlands (Kentula, pers. comm.). The general conclusion is that the strucutral attributes of created wetlands are not as planned.

With recognition of these problems, there is increasing demand for improved planning, better project implementation, and monitoring of vegetation establishment. Still, there is continuing concern that constructed wetlands do not replace lost functional values and that we don't know how to correct the situation.

Two representatives of the U.S. Fish and Wildlife Service (Holmberg and Misso 1986, p. 11) acknowledge that "...large-scale artificial wetland creation as a viable method of replacing functional natural wetlands has not been documented." Likewise, the Environmental Protection Agency has a "conservative policy" on mitigation because of the scientific uncertainty associated with man-made wetlands (Ciupek 1986). Quammen (1986) reviewed several reports on mitigation and concluded "...that compliance is low and that the effectiveness of restoration to compensate for wetland losses cannot yet be determined."

The question is, what should be done? Golet (1986) recommends that regulatory agencies take a conservative stance on mitigation by rejecting proposals where loss of wetland is avoidable and reject proposals that result in net losses of wetland area. Instead, he favors protecting wetlands in their natural state and requiring any replacement wetlands to "recreate, as nearly as possible, the original wetlands in terms of size, type, geographic location, and setting...." Quammen (1986) recommends clearly stated objectives and design criteria for mitigation wetlands, with monitoring to be conducted and reported. Good (1987, p. 107) calls for "More systematic monitoring and evaluation of existing and soon-to-be-constructed mitigation projects...to expand our knowledge base. Experiments need to be incorporated into wetland restoration projects to evaluate our hypotheses about what does and does not work. Documentation and sharing of successes and especially failures is needed, not only among agencies, but in the published literature."

To assist in the evaluation and documentation of mitigation efforts that are already underway, and thus to improve the scientific basis for accepting or rejecting mitigation plans, we developed these recommendations for assessing how well constructed wetlands replace the functions of natural wetlands in southern California. While other assessment protocols have been developed (e.g., Habitat Evaluation Program of the US FWS; Adamus and Stockwell 1983; Adamus et al. 1987), there is urgent need to tailor procedures to the special environments of this arid region. Just as the list of wetland functions needs to be refined for the region's rare and threatened habitats, so do the procedures for evaluating their presence.

2. Rationale for functional assessment

What constitutes "successful" mitigation is highly controversial (Strickland 1986, SF BCDC 1988, Zedler 1988a,b), and judgments may differ depending on the evaluation criteria used and the reference data or comparison sites. If mitigation projects are erroneously judged to be successful, natural resources may be lost irreversibly, and threatened species may move closer to extinction. However, the risks are less if mitigation projects are mistakenly judged to be failures (i.e., if wetland functions can be replaced but we don't realize it). It is better to be cautious than to assume success prematurely if we are to prevent the net loss of in-kind resources, as required by mitigation policy (US FWS 1988). Conclusive evidence that constructed wetlands can replace natural wetlands would facilitate restoration and enhancement efforts. Conclusive evidence that certain mitigation procedures cannot maintain resources would reduce economic losses by helping to resolve conflicts early in the planning process.

Disillusionment with the functioning of restored marshes has led to stricter goals for mitigation and enhancement projects in coastal wetlands and the need for clearer methods of assessing "success." Functional, in addition to structural, attributes of salt marshes are being emphasized by both researchers and resource agencies in southern California. A marsh constructed by California Department of Transportation as mitigation for highway expansion must provide self-sustaining populations of plants (including an endangered annual hemiparasite), vegetative cover that has resilience and nitrogen-fixing capability, and food chain support functions for wetland-dependent endangered birds. The assessment protocol includes the analysis of soil processes (decomposition, organic matter accumulation, nitrogen fixation, nitrogen trapping, sulfide accumulation) and long-term monitoring of transplant expansion rates, reproductive potential of the endangered plant, and habitat-specific uses by invertebrates, fishes, and birds. In this coastal region with many endangered species, functional values of lost wetlands must be replaced in accordance with new and stricter methods of assessing "success."

3. Objectives of assessment

Scientists and managers recognize three classes of functional values for the nation's wetlands (Adamus and Stockwell 1983): hydrologic functions (e.g., flood peak reduction, shoreline stabilization, groundwater recharge), water quality improvement (sediment accretion, nutrient uptake), and food chain support (habitat and food, especially for commercially important fish and shellfish and for esthetically appreciated birds). Many of these functions are less important in southern California's salt marshes (National Wetland Technical Council 1985, Onuf et al. 1978), because they are small in size (with little area to slow flood waters), and located on the coast and not upstream of potable water supplies.

For southern California, the decline in quantity and quality of our coastal wetlands has increased the importance of providing habitat for communities of organisms that can live nowhere else; several wetland species, including plants, invertebrates, fishes, birds, and mammals, are threatened with extinction. Although providing habitat for rare and endangered species is often the management goal, mitigators have not been required to guarantee their presence. Rather, their "habitat" has been the goal of several restoration and mitigation projects. In addition, there is considerable habitat value for migratory birds that rest and feed in coastal wetlands (Onuf et al. 1978, Onuf and Quammen 1985).

The need for strict assessment criteria follows from the fact that there is so little information on how well constructed wetlands carry out the functions of their natural, southern California models. Eleven functions of wetlands are considered essential for restoration success (from Zedler et al. 1988, 1990):

a. Provision of habitat for wetland-dependent species
b. Support of food chains
c. Transformation of nutrients
d. Maintenance of plant populations
e. Resilience (ability to recover from disturbances)
f. Resistance to invasive species (plant or animal)
g. Resistance to herbivore outbreaks
h. Pollination
i. Maintenance of local gene pools
j. Access to refuges during high water
k. Accommodation of rising sea level.

a. Provision of habitat for wetland-dependent species. For a region with many rare and threatened species, this is the most valued coastal wetland function. Several species of birds, mammals, insects, and a few plant species are frequent management targets, with maintenance of the entire ecosystem recognized as an essential management goal.

For the light-footed clapper rail *(Rallus longirostris levipes)*, the lower marsh provides tall, dense cordgrass for cover and nesting sites, frequent tidal inundation to deter mammalian predators, litter in the form of weaving materials for nest construction, sufficient area for territory establishment, and access to food. In addition, the birds need higher elevation refuges during times of extreme high sea levels.

For Belding's Savannah sparrow (*Passerculus sandwichensis beldingi*) the mid-to-upper marsh provides territorial males with singing perches, females with suitable nesting sites and materials, proximity to food supplies, and disturbance buffers. Sparrows also need refuges during high sea level.

For salt marsh bird's beak *(Cordylanthus maritimus* ssp. *maritimus)* the upper salt marsh provides a regeneration niche (an annual supply of open space for germination of this annual, *sensu* Grubb 1977) and suitable host species for the seedlings to parasitize.

Adjacent upland habitats are needed to support pollinator insects.

For wandering skippers (*Panoquina errans*, a rare butterfly), the salt marsh provides dense patches of saltgrass (*Distichlis spicata*), on which the larvae are host specific.

b. Support of food chains. While Pacific Coast wetlands probably are less important to coastal fisheries than Atlantic Coast wetlands, shallow tidal creeks do provide nursery grounds for Pacific halibut (C. Onuf, US FWS, pers. comm., S. Kramer, UCSD, pers. comm.). Nordby (SDSU, pers. comm., Zedler and Nordby 1986) believes the fishery values are primarily in providing food for higher trophic levels, including terns, herons, and other fishes (e.g., halibut, diamond turbot). The endangered California least tern *(Sterna antillarum browni)* depends on shallow waters for food. Elsewhere in the nation, the nursery function that serves fish and shellfish populations is often the foremost management goal. The support function is perceived as follows:

Marsh plants produce organic carbon, making it available to consumers and decomposers. Epibenthic algae produce dissolved organic carbon (absorbed by invertebrate larvae) and highly digestible biomass (consumed by invertebrates and fishes). Macroalgae provide attachment sites for topsmelt (*Atherinops affinis*) eggs. Vascular plants produce organic material for decomposers and herbivorous insects; they also house many species of insects and spiders. Two classes of decomposers modify the vascular plants; shredders break up dead plant material, facilitating leaching and microbial growth, and microbial organisms attack particles of litter (decomposition), making the detritus more nutritious for higher consumers.

The burrowing benthic animals (clam species, ghost shrimp, polychaetes, arrow gobies) mix the sediments (bioturbation), aerating the soil, enhancing microbial activities, and disrupting the soil surface. Benthic molluscs, worms, and crustaceans consume foods produced in, and tidally transported through, the marsh. Suspension feeders filter particles from the water. Deposit feeders scrape the soil surface and deposit fecal pellets and/or middens. Fishes and birds consume algae, detritus, and invertebrates, as well as frogs and other fishes.

c. Transformation of nutrients. These activities include microbial and chemical processes controlling the concentrations of nutrients and other compounds and facilitating the biogeochemical cycling of nutrients and the flow of energy. Microbes play an important role in nutrient dynamics. Cyanobacteria (blue-green algae) and soil bacteria fix nitrogen; these supplies may be essential for plant growth during times of low nitrogen influx. Nitrification produces nitrates, which are available for plant uptake. Nitrates are also substrates for denitrification, and a high nitrification rate is central to rapid nitrogen cycling. Bacteria release nitrogen as gas (denitrification), thereby reducing con-centrations of this nutrient during times of excess influx or accumulation. Bacteria reduce sulfates to sulfides (sulfate reduction) under anaerobic conditions; this may be followed by the precipitation of iron sulfides. Sulfate reduction thus plays a significant role in organic matter decomposition; these microbial reducers are available as food, and the reduced sulfides store energy for other bacteria.

Nutrient transformations are not well known in Pacific coastal wetlands. Because plant productivity of Atlantic Coast and Gulf of Mexico salt marshes is often nitrogen limited (Odum 1988), this element has been the focus of most assessments of nutrient dynamics. Recent studies suggest that nitrogen limits plant growth and affects species interactions in southern California (Covin 1984). Nitrogen dynamics should thus be considered in evaluating the functioning of constructed marshes. At present, little is known about nitrogen transformations in southern California salt marshes. Although nitrogen fixers are present among the abundant algal mats and near plant roots (rhizosphere), it is not known if the fixed nitrogen is available to vascular plants. It is likely that processes such as nitrogen fixation and denitrification are limited by organic matter availability in the soil. If this hypothesis is true, then the presence of organic matter can determine the success of wetland restoration or creation. The techniques for evaluating those processes are not simple and must be standardized. Thus, establishing methods for measuring nutrient transformation functions is an important part of our current research program.

Nutrient dynamics are strongly linked to soil organic matter, which stores nutrients and provides organic substrates for bacteria involved in nitrogen fixation, denitrification, and the sulfur cycle. Salt marsh vegetation tends to be nitrogen limited throughout coastal environments, including southern California (Covin and Zedler 1988). Floods provide an influx of nutrients, but this source is infrequent and undependable in our Mediterranean-type climate. It is unlikely that all of the nitrogen required of marsh vegetation can be supplied by tidal waters (Winfield 1980). Hence, nitrogen fixation is important for salt marsh plants, especially for a region with hypersaline soils, where plants may require extra nitrogen to regulate water uptake--some accumulate amino acids as osmolites. The potential for nitrogen-fixation is higher where soil organic matter content is high (Zalejko 1989). The sulfur cycle is also affected by low organic matter (Cantilli 1989). Man-made wetlands are characteristically lower in soil organic matter (Lindau and Hossner 1981, Shisler and Charette 1984, Broome et al. 1986, Swift 1988, R. Langis et al. in press), so the nutrient dynamics of artificial wetlands may limit the development of vegetation.

d. Maintenance of plant populations. "Successful restoration" is often judged by the survival of transplants to a specific date. Short-term objectives, such as a requiring only the survival of transplants and not their establishment, persistence, and spread, allow contractors to be freed of responsibility, even when the intent of restoration is not fulfilled.

A functional criterion, such as requiring self-maintaining populations, is needed. Self-maintenance requires that conditions favoring persistence, such as adequate nutrients (and processes such as nitrogen fixation) be present. Longer-term monitoring, and measurements of vegetative expansion and seed production, should be employed.

Persistence of plant populations develops through three mechanisms. The marsh substrate maintains seed banks; these are especially important for short-lived, non-rhizomatous species, e.g., bird's beak, annual pickleweed *(Salicornia bigelovii*), sea-blite *(Suaeda californica)* allowing recovery from mortality events. Perennial species may persist in part through longevity of individuals. The rhizomatous species persist belowgound, even though individual stems may die each year. Vegetatively reproducing species maintain potential for expansion of clones.

e. Resilience. The region's high environmental variability leads to the need for constructed wetlands to be resilient, i.e., able to recover following extreme events, as well as human disturbances. Coastal wetlands are subjected to natural hydrologic alterations, such as flooding and closure to tidal action. Human impacts include street runoff, inflows of fertilizers, pesticides and toxic wastes, mechanical damage to vegetation, increased sedimentation, and encroachment by pets.

Alterations to a wetland's hydrologic regime affects nearly every aspect of ecosystem structure and function. In 1984, Tijuana Estuary was closed to tidal flushing for 8 months--a result of long-term human disturbances to the barrier dune and storm-induced overwashing in 1983. The estuarine marsh became dry and extremely hypersaline (Zedler and Nordby 1986). The clapper rail population neared extinction as its food and vegetative cover died out. After 5 years, recovery is not yet complete--the annual pickleweed persisted only in a single, tiny patch; cordgrass has not regained all of its former distributional range; many of the macroinvertebrates failed to recolonize; the rail population has only half the number of nesting pairs as in pre-closure years.

From this whole-system catastrophe, we learned that our remnant wetlands have limited resilience. Because regional biodiversity is already in jeopardy, it is essential that any artificial wetlands be planned with that biogeographical consideration in mind, Persistence of species, and maintenance of the gene pool is a regional function. The extremes that individual wetlands experience may eliminate entire populations; for that system to recover, propagules must arrive from another. The presence of species in nearby wetlands should speed recovery by increasing the availability of propagules. Such refuges are especially important for plant species that do not develop persistent seed banks (e.g., *Salicornia bigelovii).*

In order for the fish and benthic macroinvertebrate communities of a constructed wetland to be resilient, there must be access to larvae; thus the site must have hydrologic connections with other systems in the area. For the insect community to be resilient, constructed marshes should not be too isolated. In a region where extreme and sometimes catastrophic events can't be avoided, it is essential that the wetlands retain connections with other systems to aid recovery of sensitive populations.

The accumulation of organic matter in soils also plays an important role in conferring resilience. Organic soils retain moisture that may help plants survive periods of infrequent or brief tidal inundation. Organic soils that are wetted by seawater retain some buffering capacity; they may prevent the development of acid sulfate soils upon exposure of sulfide-rich (H_2S, FeS, FeS_2) sediments. The organic matter provides an energy source for nitrogen fixers and decomposers.

f. Resistance to invasive species (plant or animal). The continual threats of disturbance to topography and salinity lead to the need for constructed wetlands to resist invasive species (exotic to the region or alien to the habitat). Once established, many invasive species can persist. A single season of altered hydrology may be enough to allow establishment and long-term presence of unwanted species. The cattail (*Typha domingensis*) is a local native of brackish marshes that invaded the San Diego River salt marsh when the period of freshwater inflow was artificially prolonged by reservoir discharges in 1980 (Zedler and Beare 1986). It was still present in 1990, even though current salinities would not allow its establishment. Furthermore, a moderate increase in freshwater inflow at this time could stimulate vegetative expansion now that the species is present in the marsh.

The hydrologic regime (salinities and soil moisture regimes) usually precludes establishment of species foreign to the salt marsh, such as cattails (*Typha latifolia*) and bulrushes (*Scirpus* spp.). The mature marsh sod and vegetation canopy function to reduce seedling survival of exotic plants such as rabbit-foot grass (*Polypogon monspeliensis*) and brass buttons (*Cotula coronopifolia*). Constructed salt marshes may lack sufficient vegetative cover to prevent invasions. If wetlands are constructed in urban areas, freshwater inflows may be augmented by street runoff, and soil salinity regimes may favor germination and seedling establishment of brackish marsh species. If the hydrologic regime cannot be corrected, weed control measures must be planned and implemented as aliens appear. Most invaders will be easiest to manage if individuals are hand pulled early in the invasion process.

Exotic animals are also of concern. Mosquitofish *(Gambusia affinis)* and sailfin molly *(Poecilia latipinna)* have been recorded from coastal wetlands in the San Diego area. The yellowfin goby *(Acanthogobius flavimanus)* is a current threat that appears to be moving south along the California coast with extreme rapidity. A benthic mussel from Japan *(Musculus senhousia)* has invaded subtidal habitats of both Mission Bay and San Diego Bay, where it seems to displace native bivalves (D. Dexter, SDSU, pers. comm.). In studies along San Diego Bay, comparing a constructed wetland with a natural marsh remnant, S. Rutherford trapped many individuals of the exotic mussel in the artificial marsh, but found only one in the natural system. Once established, such exotics may preclude development of the native fauna.

We do not know what defense mechanisms natural wetlands may have to prevent or slow invasions of exotic species. Where native species are already abundant, there may be insufficient food or space for invaders. The salinity regime may prevent survival or settling of larvae that are alien to the salt marsh, such as mosquitofish. A dense community of benthic organisms may filter exotic larvae from the water column.

g. Resistance to herbivore outbreaks. In native wetlands, insect herbivores are diverse and abundant, but population outbreaks are uncommon. Native predators, including birds, carnivorous insects, and parasitic wasps, keep herbivore populations in check. Where such predators are lacking, herbivore populations may go out of control. Herbivores also respond to the nutrient status of plants. Where nutrient-rich wastewaters flow into coastal wetlands, the nitrogen status of the salt marsh plants improves, and insect grazer populations may expand before predators are able to crop the increased prey base.

When the functions that confer resistence to outbreaks are lacking and herbivores get out of control, the vegetation is negatively affected. A cordgrass *(Spartina foliosa)* planting on a dredge spoil island in San Diego Bay looked like a "successful" marsh for the first three years; then, a population of scale insects (*Haliaspis spartina*) irrupted and nearly decimated the vegetation. Kathy Williams (SDSU, pers. comm.) attributes the outbreak to a lack of predators. In natural marshes a coccinelid beetle (*Coleomegilla fuscilabris*) appears to control outbreaks--under experimental conditions, it feeds voraciously on scales. Neither parasites nor predators were deliberately introduced to the marsh, and they did not become established on their own. This was our first experience with an insect outbreak, and no one anticipated it. We learned that resistance to defoliating herbivores is an important function of natural marshes. Because it may take pest species several years to cause noticeable mortality or defoliation, the success of a marsh construction project cannot be determined without long-term study.

h. Pollination. While many salt marsh plants lack showy flowers and are wind pollinated, a few species rely on insects. The annual salt marsh bird's beak (*Cordylanthus maritimus* ssp. *maritimus*) is pollinated by bees and flies, on which this endangered species relies for successful seed set. For such insects to be present, their nesting habitat must be also be available nearby--for the 5 species that potentially pollinate bird's beak (Lincoln 1985), salt flats and higher ground with mammal burrows are the likely nesting sites.

Brian Fink successfully transplanted the bird's beak to a restored higher marsh area at Sweetwater Marsh in 1990; however, flowers did not produce seeds. He attributed the population failure to lack of habitat for the pollinators (Fink, SDSU, pers. comm.).

i. Maintenance of local gene pools. A species may be maintained through transplantation or other means of artificial propagation, but its genetic integrity may be modified in the process. Nearly every author for EPA's regional restoration reviews (Kusler and Kentula 1989) expressed concern that local gene pools be maintained in nature.

The gene pool is threatened in at least two ways--when local genetic diversity is not salvaged and when alien material is brought in from other regions. In the first case, the failure to retain a broad representation of local plants or animals may result in a wetland with low genetic diversity, with reduced material for natural selection. In the second case, local gene pools may be reduced through competitive exclusion that would not normally occur.

Genetic integrity is not only important for its own sake. Long-term maintenance of the habitat may be at risk. A plan to transplant cordgrass from San Francisco Bay to San Diego Bay, 500 miles to the south, was criticized by PERL biologists who argued that the northern populations might be less tolerant of high temperatures and hypersalinity. An extreme event could wipe out the imported material, perhaps years after establishment. A local source of cordgrass that was destined for destruction was identified and used instead.

j. Access to refuges during high water. California's Coastal Act requires a 100-foot buffer around wetlands to reduce disturbances to the protected ecosystem, although some have considered bike trails and parking lots compatible uses within the buffer zone. The importance of having a functional refuge is evident during extreme high waters, when clapper rails and other animals need to escape their flooded habitat. Storms can raise water levels well above predicted tidal heights, and if storms coincide with the highest spring tides of the year (as in January 1988), then the entire intertidal marsh will be flooded with two feet of water or more. Where a habitat refuge is lacking, survival through extreme events is uncertain. While the refuge function of buffers needs further quantification, it is clear that buffers are necessary during storm periods and for future intertidal habitat as sea level rises.

k. Accommodation of rising sea level. Park et al. (1989) estimate that a one-meter rise in sea level would eliminate 65% of the coastal marshes and swamps of the contiguous US, because the upward migration of wetland ecosystems is restricted by coastal development. Such a rise is probable by the year 2100. In southern California, even a half meter rise, would wipe out closer to 100% of the remaining wetlands, because the upland transitional habitats needed to support the rising marshlands are all developed. We're probably stuck with rising sea level, so restoration planning should accommodate it. Instead of grading an entire site to intertidal elevations, transitional habitats should be left, and connections to upland topography should be maintained. As Bill Niering of Connecticut College wrote (in Kusler and Kentula 1989), "Our basic goal is to create a persistent functional wetland system. In some situations this may be more important than the creation of a specific wetland type since the present physiognomy may merely be a momentary expression of the system's vegetation potential in the future."

4. Criteria for "successful mitigation" as required in San Diego Bay

How similar should the functions of constructed and natural wetlands be before the project is considered successful mitigation? In order to insure no net loss of habitat values, it is necessary to have high expectations for wetlands constructed or modified for the purpose of mitigating losses elsewhere. In the past, standards have varied from project to project. Where standards are lax, there is greater risk that efforts to retain or improve upon wetland values will also be inadequate.

High standards are necessary if projects are to comply with agency policies, and detailed criteria for assessing success are needed to document in detail what is happening on the restoration site. In addition, scientific studies should be included to identify what causal factors are responsible. If, for example, nutrient concentrations are unusually low, an understanding of the reasons will help in identifying corrective measures and in preventing similar problems in the future (Broome et al. 1987).

Obviously, for a wetland designed to provide habitat for use by endangered birds, judgments of success should include the presence of those populations and perhaps their self-maintenance. Annual censusing of rare and endangered birds is needed plus 5- and 10-year follow-up studies of the plant and animal communities and ecosystem functions that develop on site.

Special attention should be paid to problem species. At least three categories have been identified in constructed wetlands: invasive exotics (e.g., rabbit-foot grass), invasive natives that are alien to the desired wetland community (e.g., cattails), and resident natives that undergo population irruptions not seen in natural wetlands (e.g., scale insects on cordgrass). Annual surveillance will be needed for problem species, so that corrective measures can be initiated early enough to be effective.

Finally, we recommend that an objective scientific panel review the sampling or monitoring program used to judge compliance/success. For the Caltrans Connector Marsh, the US Fish and Wildlife Service anticipated the need for interim reviews of the mitigation program. The federal agencies and contractors must meet with the Service annually to review project status and recommend any remedial actions (US FWS 1988, p. 23).

In a recent decision, the US Fish and Wildlife Service prescribed detailed objectives, including functional attributes, for evaluating the success of the mitigation project at the Connector Marsh, constructed by Caltrans in Chula Vista. Three endangered species occur in the affected area, the light-footed clapper rail, the California least tern, and the salt marsh bird's beak. When the Sierra Club charged the federal agencies with failure to enforce the original mitigation requirements (primarily a transfer of land from private to public ownership, subsequently ordered by the court; Thompson 1988), it became possible to revise and update the "Section 7 Consultation" under the Endangered Species Act, strengthening the requirements both for mitigation and for judging its success.

The new requirements are set forth in the "Biological Opinion" (US FWS 1988) for the Combined Sweetwater Flood Control and Freeway Project. "The Service shall deem that the wetland creation and modification projects are successful on showing that the channels and emergent wetlands provide suitable, functional habitats for the California least tern and light-footed clapper rail, and the emergent wetlands are also vegetated by patches of salt marsh bird's beak and 75 percent of the native species currently occurring in the Sweetwater River Wetlands Complex." (*ibid.*, p. 23) The Service went beyond this general goal and provided specific criteria as follows:

Channels need to provide "suitable habitat for the California least tern and the light-footed clapper rail." Forage items need to be present for 2 years at levels of 75 percent of the density and diversity of the prey base in comparable habitats with the Sweetwater River Wetlands Complex (SRWC). Wetland habitat for rails must include 7 home ranges for 2 years, each with non-overlapping areas of 2-4 acres, including low, middle, and high marsh. Lower marsh in each home range must have at least 1 patch of cordgrass of 60-80 cm height and 90-100% cover that is 90-100 m^2 in size and able to maintain itself (i.e., in place for 3 yr and with N-fixation). The middle marsh must have $\geq$70% cover with 75% of the typical native species at SRWC. The high marsh must have $\leq$20% cover of weedy species and maintain 5 separate patches (of 1 m^2 and at least 10 m part, with $\geq$20 plants/patch) of bird's beak that are self-sustaining (stable or increasing in number and area) for 3 yr. An attempt should be made to include high saltmarsh berms near areas of low saltmarsh.

These new requirements represent a major advance in agency expectations for successful mitigation. Their effectiveness in insuring the replacement of in-kind habitat will be evaluated along with the constructed habitat.

5. Reference wetlands and reference data sets

In order to compare constructed wetlands with natural ecosystems, there must be data on the structure and functioning of representative natural wetlands. Because most southern California wetlands have been disturbed, there are no pristine wetland examples. Even in nearby Baja California, where coastal development is less extensive, there are few examples of undisturbed habitat. Selection of wetlands to serve as models for restoration must therefore be based on an understanding of how disturbance affects wetland ecosystems. Two additional features of Southern California wetlands make it difficult to characterize the reference wetland or restoration target. The first is spatial heterogeneity within and between wetlands; the second is interannual variability, due to both natural and man-caused events.

The problem of spatial variability. No two wetlands are identical in species lists, distributions, or rates of various processes. Yet the spatial variability among the region's wetlands is more of an aid than a hindrance in understanding cause-effect relationships. Current knowledge of wetland structure and function comes from long-term experience and study in several wetlands, with each site contributing unique information.

Four main study sites have contributed to our understanding, with the longest period of record at Tijuana Estuary (Table II.1). The results of these studies indicate that no one wetland can serve as a reference site for any man-made habitat. Information from constructed wetlands needs to be compared with the total available information, preferably by a panel of scientists familiar with the study sites, to insure thorough, objective evaluation. Such a process has been initiated by Caltrans, involving scientists from the Pacific Estuarine

Research Laboratory and resource agency staff to help assess wetland functions at the Connector Marsh.

The problem of temporal variability. It has taken several years to understand just a few of the functions of the Tijuana Estuary salt marsh, largely because of the high interannual variability and the importance of extreme events in triggering changes in the vegetation. Ten years of data on plant distributions and abundances are still inadequate, because we have not witnessed all types of extreme events or human impacts. Our knowledge of vegetation control functions includes relationships with salinity-inundation regimes, but is weak concerning interactions with nutrients, soil redox conditions, heavy metals and other toxic materials. Yet this is the best data set available, and it serves to caution managers that high interannual variability may be the rule, rather than the exception, for the region's wetlands.

It is thus apparent that short-term assessments must document trends, rather than "average states." Monitoring programs must take more than "snapshots" of the system under study; time series are needed to determine the direction of developments. Rather than focusing on one-time measures of transplant survival (often the only requirement for monitoring mitigation sites), it is more important to assess the ability of the system to respond to changing environments (e.g., expansion following winter streamflow and low soil salinities) and the likelihood that populations can be maintained through environmental extremes (e.g., measures of seed production; size of seed bank), to identify any major shifts in the community (e.g., loss of desired plant species), and to develop corrective measures for problems that are seen (e.g., eradication of exotic invaders before they become widespread).

The region's wetland resources have not been censused in detail. The US Fish and Wildlife Service (National Wetlands Inventory) has prepared draft maps of habitat types in southern California coastal wetlands, based on 1985 aerial photography. However, their habitat classifications have not been checked in the field. Sharon Lockhart used the FWS draft maps to estimate the areas of 23 coastal wetlands in San Diego County. She outlined each of the polygons with a planimeter to obtain areas, and summarized the acreage of each habitat type and each wetland. Acreages included areas between the ocean inlet and the first major break in habitat toward the east. Freeway 5 was the limit for wetlands in San Diego and Mission Bays; El Camino Real was the limit for Los Peñasquitos Lagoon north to Buena Vista Lagoon. The FWS habitat classifications were combined into 5 habitat types: bay, channel, salt marsh, brackish/fresh marsh, impounded waters, and other habitats (riverine and unidentified polygons).

Excluding the 5,420 ha of subtidal or bay habitat, this survey indicates 756 ha of salt marsh, 832 ha of brackish and fresh marsh, 495 ha of impounded wetlands, 163 ha of channels, and 32 ha of other wetland habitat types in San Diego County (Figure II.1). However, it is likely that these are underestimates because many wetland areas may not be distinct on aerial photos. Comparative, field based data for wetlands at Tijuana Estuary were obtained in a PERL survey, which included extensive field sampling over large areas of disturbed wetland. The map by PERL ecologists indicated much larger areas of wetland at Tijuana Estuary than the FWS map (556.3 ha compared to 284.1 ha). For Tijuana Estuary, the FWS draft inventory appeared to be a 50% underestimate of wetland area.

For Tijuana Estuary, the areas of habitat from more detailed mapping, including ground truthing, are given in Figure II.2. The areas were obtained using the SDSU Geographic Information System.

Table II.1. A list of sites and types of data currently gathered by scientists associated with the Pacific Estuarine Research Laboratory. While there are additional data for these and other sites, only sampling programs that use comparable methods are listed. TE = Tijuana Estuary; SDR = San Diego River Marsh; LPL = Los Penasquitos Lagoon; CM = Connector Marsh and PC = Paradise Creek, both on San Diego Bay. Similar vegetation and salinity data have been obtained at Estero de Punta Banda, Baja California, by Silvia Ibarra-Obando at CICESE, Ensenada, B.C.

Location:	TE	SDR	LPL	CCM	PC
Soil salinity data	X	X	X	X	X
Streamflow data (USGS)	X	X	X		
Lower salt marsh veg. data	Sept.	July	Sept.	summer	summer
Number of stations	102	250	86	64	7
Number of years	10	6	2	3	3
Nutrient dynamics data				qtrly	qtrly
Number of years			5	3	3
Upper salt marsh data	Sept.		Sept.		
Number of stations	115		12		
Number of years	4		5		
Fish and invertebrate data	qtrly		qtrly		
Number of stations	4-6		3		
Number of years	4		5		

Figure II.1. Comparison of habitat areas in 23 wetlands of San Diego County. Areas exclude subtidal bay habitats.

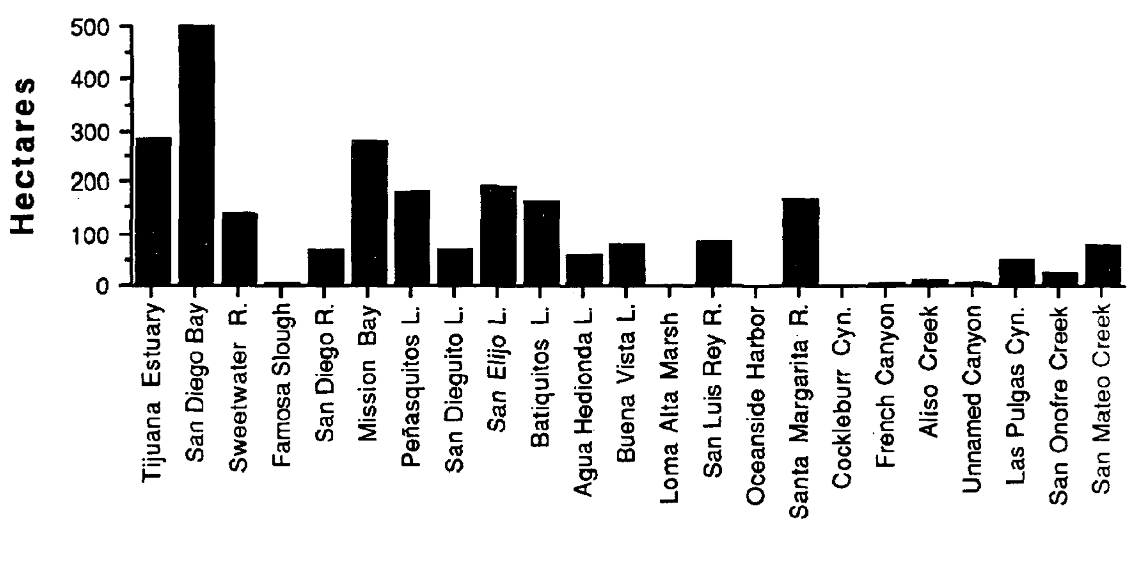
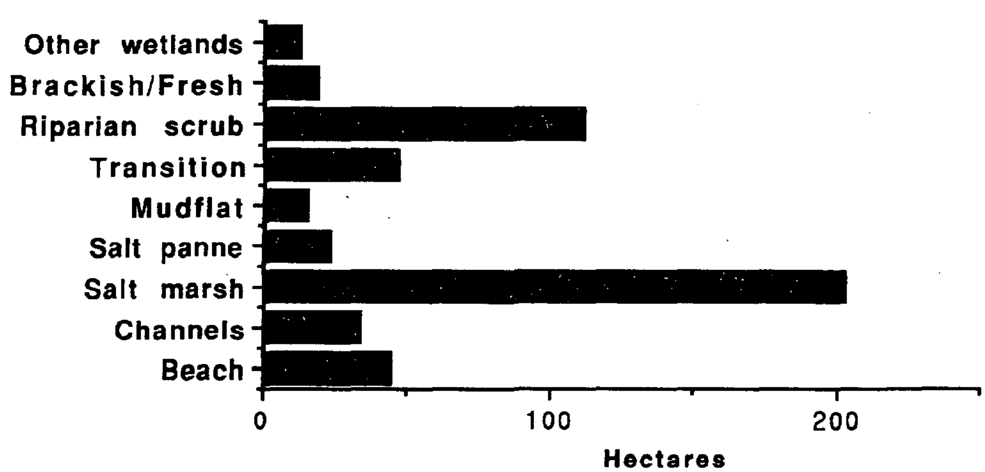


Table II.2. San Diego County wetland habitats, determined from US FWS draft maps of the National Wetland Inventory by S. Lockhart. Data are hectares (1 ha = 2.471 ac). Br/Fr = Brackish/Freshwater.

Location	Bay	Channel	Saline Marsh	Br/Fr Marsh	Impounded Waters	Other
Tijuana Estuary	0.0	223.6	171.1	88.1	0.0	1.4
San Diego Bay	4483.2	.2	11.0	23.3	430.9	0.0
Sweetwater	0.0	1.9	116.1	18.6	0.0	1.0
Famosa Slough	0.0	2.6	1.5	0.0	0.0	0.0
San Diego River	62.6	0.0	47.1	19.8	0.0	1.1
Mission Bay	620.1	0.0	35.3	39.9	0.0	0.0
Los Peñasquitos	0.0	3.1	130.7	50.0	0.0	0.0
San Dieguito	30.0	0.0	28.7	36.7	0.0	0.8
San Elijo	0.0	31.6	87.6	74.2	0.0	0.0
Batiquitos	42.7	99.9	8.6	52.7	0.0	0.0
Aqua Hedionda	105.9	0.1	31.4	26.9	0.0	0.0
Buena Vista	0.0	0.0	0.0	15.6	64.6	0.0
Loma Alta	0.0	0.0	0.0	0.7	0.0	0.0
San Luis Rey	0.0	0.7	0.0	82.6	0.0	2.0
Oceanside Harbor	85.3	0.0	0.2	0.0	0.0	0.0
Santa Margarita	30.4	0.0	87.3	79.0	0.0	0.0
Cockleburr Cyn	0.0	0.0	0.0	2.0	0.0	0.0
French Canyon	0.0	0.0	0.0	6.3	0.0	0.0
Aliso Creek	0.0	0.0	0.0	10.2	0.0	0.0
Unnamed Canyon	0.0	0.0	0.0	3.8	0.0	0.0
Las Pulgas Cyn	0.0	0.0	0.0	46.8	0.0	0.6
San Onofre Creek	0.0	0.0	0.0	8.8	0.0	0.0
San Mateo Creek	0.0	0.0	0.0	56.3	0.0	32.2

Figure II.2. Area of several wetland habitat types at Tijuana Estuary, as determined by PERL from detailed ground truthing of the 1986 aerial photo.



6. How should major restoration programs be undertaken?

It is recommended that major restoration projects, especially those involving techniques or habitat types for which there are no previous examples, begin with an experiment to test methods of restoration on the site. A small effort in planting vegetation or modifying the topography in various configurations, followed by evaluation over a growing season, will help rule out techniques that don't work in the short-term and will help identify promising approaches that can be developed further .

All projects should be continually evaluated with a long-term functional assessment program. An adaptive management approach will improve the restoration program through time; that is, collection of information about system development, identification of problems, and experimentation with new restoration techniques, will provide the information necessary to incorporate corrective measures.

The Tijuana Estuary Tidal Restoration Program provides a good example. A plan to restore full tidal flushing to about 200 ha (500 ac) of the estuary was developed by two hydrologists, Phil Williams and Mitch Swanson (1987). Review of the initial, "minimum dredging plan" by PERL biologists indicated that some areas where dredging was proposed were critical habitat for sensitive species. Alternative sites for new channels and intertidal marsh habitats were then sought, and a compromise was developed to reduce biological impacts but require a more costly dredging program. The revised plans will soon be available as an Environmental Impact Report/Statement (EIR/EIS for Tidal Restoration of Tijuana Estuary, prepared for the California State Coastal Conservancy). The EIR/EIS proposes a low-cost experimental phase to precede major dredging and grading. Two major questions will be answered, in sequence, using replicate experimental marshes: First, how will different degrees of tidal influence affect pickleweed marsh? Second, how important is topographic complexity to wetland functioning--should small tidal creeks be excavated within the marsh plain to provide habitat for a diverse macrobenthos and food chain?

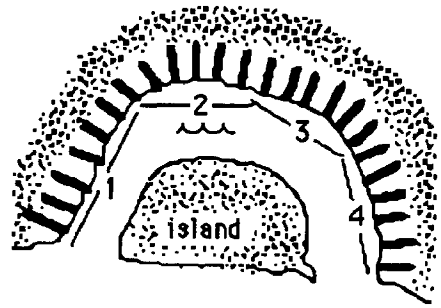
To answer the first question, 24 small tidal marshes (1x10-m mesocosms; Figure II.3) will be constructed adjacent to an existing tidal channel. Each will be assigned to one of 6 treatments (4x replication). Soil and vegetation attributes (including both vascular plant and algal producers) will be followed through one growing season. Study of the ecosystem responses will determine the degree of tidal flushing and freshwater inflow needed to manage the existing pickleweed ecosystem and to construct new pickleweed marshes.

To answer the second question, a 20-acre area adjacent to an estuarine channel will be graded down to approximately 4.5 ft MLLW for development as an experimental intertidal marsh. Replicate subareas (n=3) will be constructed with and without tidal creeks. Results from this experiment (vegetation and soil development, invertebrate colonization, and bird use) will guide implementation of later phases of the project. Since funds have not been identified for the larger restoration program, it is likely that several years of data will be available by the time results are needed. Meanwhile, information will add to the technical information base and guide restoration projects elsewhere.

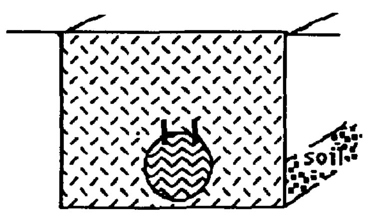
Figure II.3.

Map of 24 tidal mesocosms at Tijuana Estuary. Each of the experimental blocks (1-4) includes 6 hydrologic treatments:

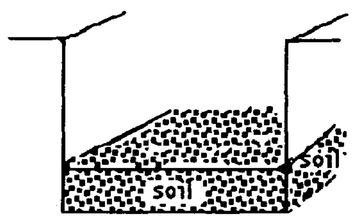
3 tidal flushing regimes, each with and without freshwater inflow.



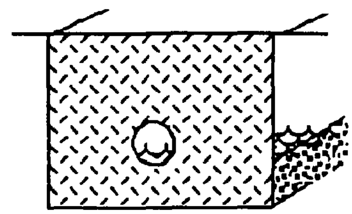
Reduced tidal flushing | **Full tidal flushing** | **Prolonged flooding**



Tidal influence restricted by one-way flap gate that reduces inflow, but allows drainage

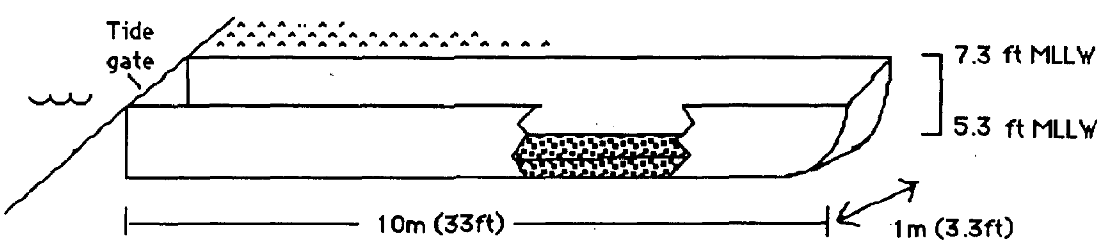


Daily tidal inundation and drainage

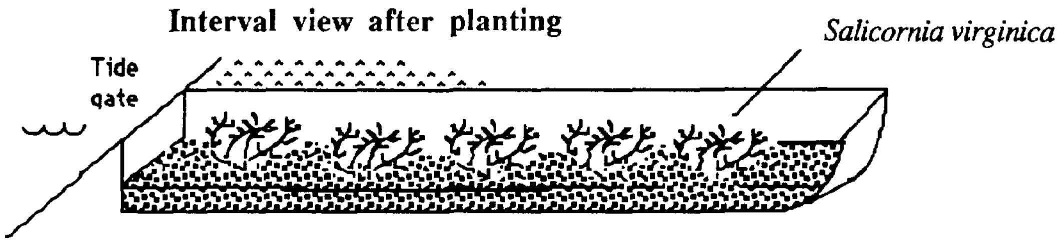


Daily tidal inflow, drainage impeded by height of outlet

View of tidal mesocosm construction plan



Interval view after planting



III. Case Study: Sweetwater River Wetlands Complex

Our research site is within the largest and most controversial (Thompson 1988) mitigation project on San Diego Bay. The project combines highway widening, construction of a new freeway interchange, and excavation of a flood control channel. Mitigation for lost wetland habitat is being carried out by the California Department of Transportation (Caltrans) and includes construction and enhancement of intertidal salt marsh habitat (cf. US ACE 1982, Swift 1988) within the City of Chula Vista (32º10'N, 117º 10'W). The mitigation marshes include the "Connector Marsh," which was constructed as a hydrologic link between Paradise Creek and the Sweetwater Marsh, and Marisma de Nación, a 17-acre marsh excavated from the "D Street fill" in early 1990. The marsh, channel, and creek habitats are within the range of three birds that are on the federal endangered species list: the light-footed clapper rail (*Rallus longirostris levipes*), the California brown pelican (*Pelecanus occidentalis*), and the California least tern (*Sterna antillarum browni*) and one endangered plant, the salt marsh bird's beak (*Cordylanthus maritimus* ssp. *maritimus*). Mitigation marshes at this site must provide functional habitat for the rail, the tern, and the bird's beak (US FWS 1988).

While the assessment study is not yet complete, results are available for soils, nutrients, vegetation, and epibenthic invertebrates (Langis et al. in press; Zedler and Langis in press). Comparisons of fishes, channel benthos, and birds are in progress.

The work has been sponsored by Caltrans and NOAA, National Sea Grant College Program, Department of Commerce, under grant number NA85AA-D-SG140, project number R/CZ-82, through the California Sea Grant College program, with matching funds from the California State Resources Agency.

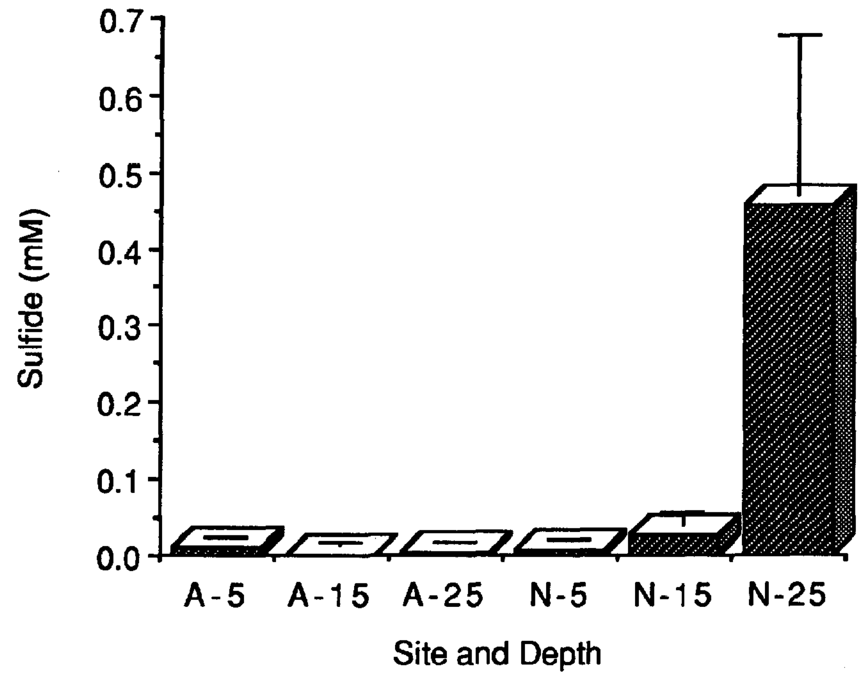
Free sulfide in natural and constructed salt marshes (from Cantilli et al. 1989 and Cantilli 1989). The presence of only trace amounts of free sulfide in a man-made marsh on San Diego Bay suggests that its biogeochemical functioning is not equivalent to that of a natural salt marsh.

Sediments in the artificial system and an adjacent natural marsh were sampled near the seaward edge of cordgrass (*Spartina foliosa*) growth. Concentrations of free sulfide (H_2S, HS) in the natural marsh were significantly greater than in the constructed marsh (Figure III.1), with levels up to 3 mM/L at 25-cm depth. Further, data on cordgrass aerial biomass from marshes at San Diego Bay and Tijuana Estuary suggest that free sulfide is not directly phytotoxic. Sulfide is a product of sulfate reduction, a process that typically dominates anaerobic decomposition and carbon cycling in salt marshes. This process may be limited by low sedimentary organic matter content in the man-made marsh (Table III.1). The near-absence of sulfate reduction in the artificial marsh may affect energy flow and export, as well as the retention of heavy metals and anthropogenic sulfur.

Table III.1. Means of percent organic carbon in marsh soils at depths of 0-5 cm and 5-10 cm. Values were obtained by weight loss (dry wt) on ignition at 700° C for one hour.

Depth	Natural	Artificial
0-5 cm	8.14	4.33
5-10 cm	9.24	3.98

Figure III.1. Pore water concentrations of free sulfides in the artificial (A) and natural (N) marshes at depths of 5, 15, and 25 cm. Bar = 1 s.e.



Potential nitrogen inputs (N-fixation) in natural and man-made salt marshes (modified from Zalejko et al. 1989, and Langis et al. in press). Data indicate that nitrogen fixation provided more nitrogen in the natural salt marsh (Paradise Creek Marsh) than in the man-made (Connector Marsh) salt marsh located at San Diego Bay.

Using the acetylene reduction technique, potential N-fixation rates were measured in both the root zone of cordgrass *(Spartina foliosa)* and on the sediment surface in association with blue-green algae. Nitrogen fixation was measured at the same intertidal elevation in both marshes to eliminate differences due to soil moisture. Rates of N-fixation on the sediment surface were significantly higher--generally twice as high--in the natural marsh than in the man-made marsh for five of six sampling periods (Table III.2, Figure III.2). In September 1988, there was no difference between the marshes. The rates of nitrogen in the natural marsh, although greater than in the constructed marsh, were still very low by comparison with Atlantic Coast wetlands. While nitrogen-fixation at this one natural marsh may not be representative of the region, the indication is that nitrogen limits the growth of cordgrass. N-fixation is one mechanism by which nitrogen can be supplied continuously without the herbivore-stimulating effects that sometimes accompanies fertilizer treatments. Lower N-fixation rates in the constructed marsh were related to lower soil organic matter levels (see next page).

Rates of N-fixation <u>in the root zone</u> of cordgrass were often similar for the constructed and natural marshes (Table III.3). Rates were significantly higher in the natural marsh for only two out of five sampling times. Langis et al. (in press) attribute the high rates in the constructed marsh to the low nitrogen concentrations found there. Nitrogen fixation is known to shut down when nitrogen is abundant in the environment.

Significant positive correlations were found between N-fixation rates and percent organic matter and belowground biomass (Figure III.3) showing the importance of primary producers for nitrogen fixation. At this study site, the natural marsh was 2.3 times higher in above-ground plant biomass (Langis and Zedler in press). Other studies (e.g., Covin 1984) have shown that nitrogen is limiting to salt marsh plant growth. N-fixation is thus a very important process because it introduces new nitrogen into salt marsh ecosystems. A positive feedback control system is suggested--as cordgrass begins growth, organic matter becomes available to N-fixers in the rhizosphere, and N-fixation is stimulated. Increased nitrogen levels then stimulate more rapid growth of cordgrass. On the soil surface, similar interactions with microalgae may be present.

Table III.2. Rates of N-fixation in surface cores (1 cm deep) for several constructed and natural marsh sites, expressed as nmol C_2H_2/hr/m^2. Data are means (and s.e.) for 5-12 samples from the Connector and Paradise Creek marshes adjacent to San Diego Bay. Data are from Langis et al. (in press).

Date	Constructed Marshes: Islands	Constructed Marshes: North Bank	Natural Marsh
Feb. 1988	**15.0** (4.62)	**7.3** (4.62)	**33.0** (4.30)
Apr. 1988	**42.6** (22.74)	**15.6** (6.35)	**118.9** (32.28)
July 1988	**45.4** (2.16)	**47.6** (4.94)	**70.2** (9.87)
Sept. 1988	**34.7** (0.99)	**46.5** (7.61)	**34.0** (1.15)
Feb. 1989	**24.5** (2.1)	**26.8** (2.68)	**61.5** (8.70)
Mar. 1989	no data	**19.5** (0.88)	**24.7** (1.49)

Table III.3. Rates of N-fixation within the cordgrass rhizosphere (10 cm deep) for constructed and natural marshes. Data are nmol C_2H_2/hr/m^2 as in Table III.2.

Date	Constructed Marshes: Islands	Constructed Marshes: North Bank	Natural Marsh
Dec. 1987	no data	**3.0** (0.51)	**7.5** (0.82)
Feb. 1988	**105.5** (30.63)	**44.3** (16.00)	**104.3** (51.31)
Sept. 1988	**34.7** (0.99)	**46.5** (7.61)	**34.0** (1.15)
Apr. 1988	**96.2** (20.56)	**77.0** (17.60)	**161.9** (30.78)
Sept. 1988	**251.7** (20.78)	**251.8** (40.56)	**73.7** (9.54)

Figure III.2. Mean N-fixation (acetylene reduction) on the sediment surface at individual sites in the artificial and natural salt marsh during July 1988. Units are C_2H_4 nmoles/g soil per hr ($*10^{-4}$). Data are from Zalejko (1989) and Zalejko et al. (1989).

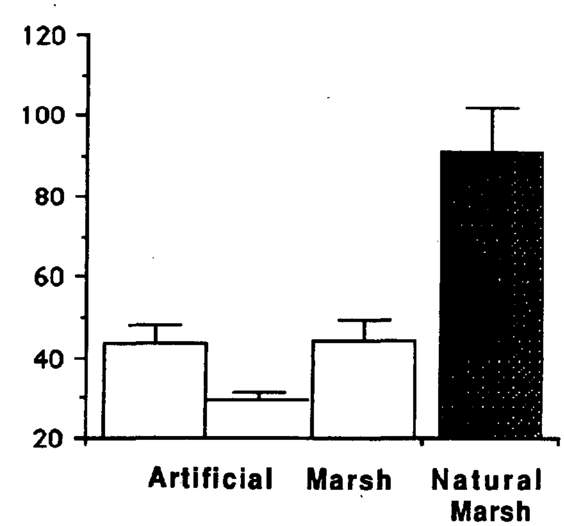
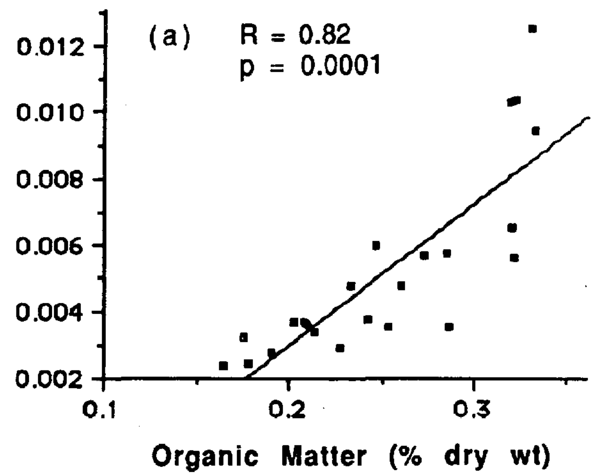
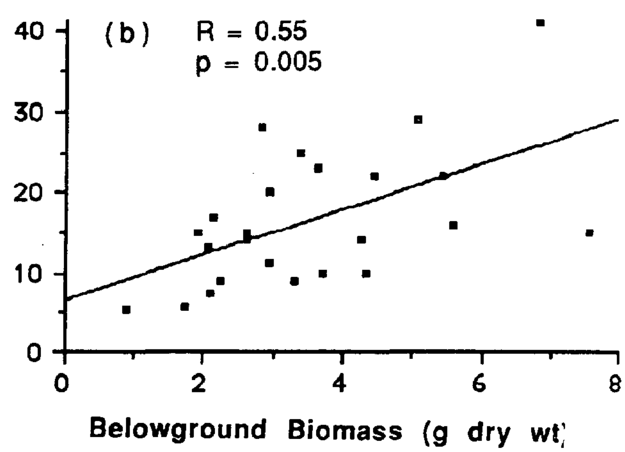


Figure III.3. Relationships between rates of N-fixation (acetylene reduction) and (a) soil organic matter content and (b) belowground biomass. Each point represents an individual measurement. Units are C_2H_4 nmoles/g soil per hr ($*10^{-4}$). Data are from Zalejko et al. (1989).





Nitrogen dynamics in natural vs. man-made salt marshes (modified from Langis and Zedler 1989, and Langis et al. in press). Nitrogen levels in sediments, pore-water and above-ground vegetation in the constructed salt marsh were compared with those in the natural Paradise Creek Marsh. Sampling sites were the same as for nitrogen fixation (above).

Nitrogen levels (ammonia, nitrate plus nitrite, and total Kjeldahl nitrogen [TKN]) were significantly higher ($P<0.01$) in the natural than in the man-made marsh in all the compartments examined. Phosphorus levels were likewise higher in the natural marsh (Figure III.4). The magnitude of differences in nitrogen concentrations between the natural reference marsh and the constructed marsh were as follows: 2.4 times higher for sediment total N and extractable NH_4; 9.4 times higher for pore-water NH_4 (Figure III.5) and 0.2 times higher for foliar nitrogen in cordgrass. Aboveground biomass of plants was much higher (Figure III.6), and percent organic matter in sediments was 2.4 times higher in the natural marsh. Total nitrogen in soils was highly correlated with percent organic matter (Figure III.7).

Clearly, nutrient dynamics in the constructed marsh are not comparable to those of the natural marsh. Since in both marshes sediment nitrogen levels (TKN) were correlated with percent soil organic matter, we believe that belowground organic matter must accumulate in soils for nitrogen levels to reach those of the natural marsh.

Through time, we expect the marsh soil to develop and for organic matter and nutrients to accumulate. However, in two years of sampling nitrogen concentrations, there was no indication of an increase. Much longer-term studies are needed to determine how rapidly wetland soils can develop and how long it may take for constructed marshes to achieve the nutrient status of natural marshes.

Figure III.4. Total nitrogen (TKN) and phosphorus determined in sediment core samples in May 1988 from the natural and man-made marshes. Differences in total nitrogen means were highly significant ($P<0.001$) while total phosphorus values were not. The low N/P ratios are indicative of nitrogen limitation. Bar = 1 s.e.

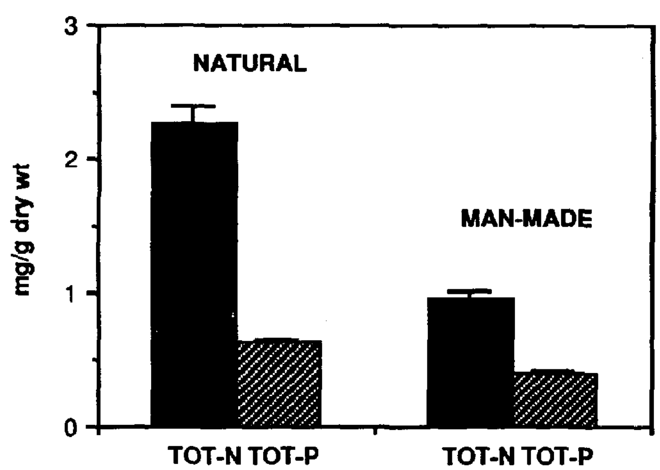


Figure III.5. Ammonium levels of pore-water samples collected September 16 and November 15, 1988. Differences between sites were highly significant ($P<0.001$). Bar = 1 s.e.

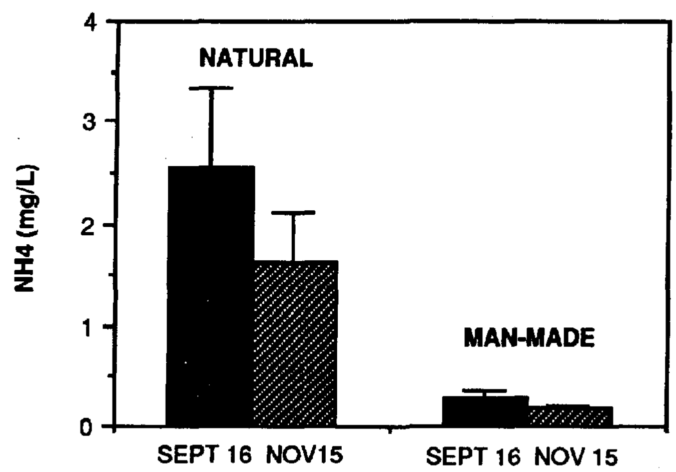


Figure III.6. Above-ground biomass of cordgrass collected July 15, 1988 from 0.25-m^2 quadrats. Differences between sites were highly significant ($P<0.001$). Bar = 1 s.e.

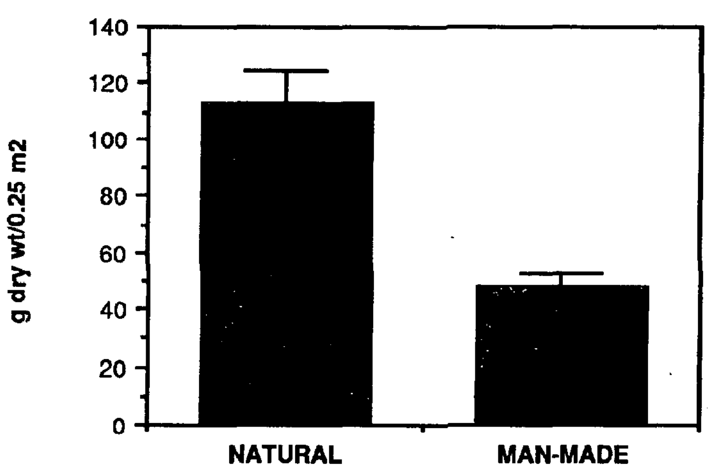
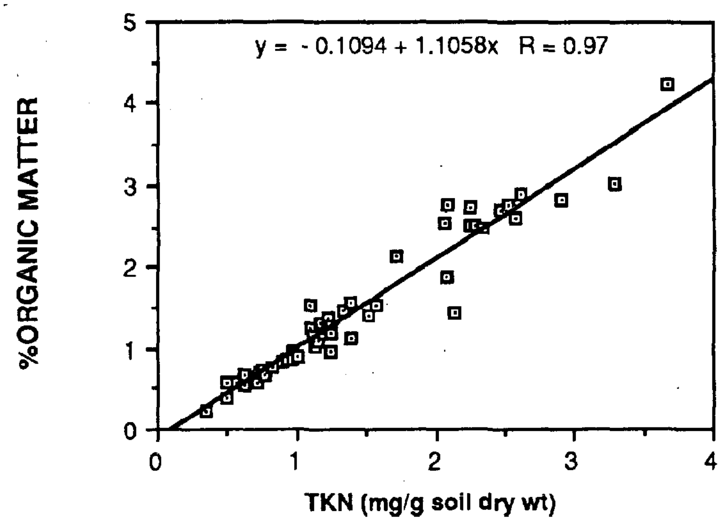
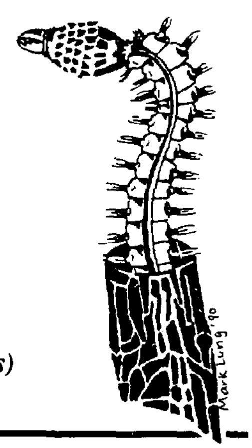


Figure III.7. Regression of sediment total nitrogen (TKN) over percent organic matter.



Epibenthic invertebrate distributions in natural vs. man-made salt marshes (modified from Rutherford and Zedler 1989; details in Rutherford 1989). There were significantly more individuals (2-4x) and more invertebrate species in a natural marsh at San Diego Bay than in the 4-yr-old man-made marsh, comparing the same low-marsh elevations (Figures III.8-III.9). All animals were trapped in litterbags filled with dried cordgrass. The most abundant species was a larval Dipteran, *Pericoma* sp., which was significantly more abundant (up to 9x) in the natural marsh (ANOVA, $p < 0.05$). An anemone, *Diadumene franciscana,* was found only in the natural marsh. In the man-made marsh, there were significantly more *Hemigrapsus* crabs in all sites sampled. In the natural marsh (at ca. 0.3 m above MSL), areas with 80-100% cover of cordgrass supported twice as many invertebrates as areas with 0-20% cover. At an elevation of 0.5 m above MSL, the numbers of species and individuals were similar for areas with high cover at the two marshes.

After four years, this marsh had not developed its natural food chain support function. As cordgrass cover increases, it should facilitate recruitment of the invertebrate community; however, it is not yet clear whether the constructed marsh provides the required quantities and qualities of food for native invertebrate populations to develop their natural abundances and to persist in perpetuity.



Polychaete worm
(Nephthys caecoides)

Figure III.8. Mean number of individuals per litterbag in areas of low elevation with high (80-100%) vs. low (0-20%) cover of cordgrass vs. high elevation with high cover.

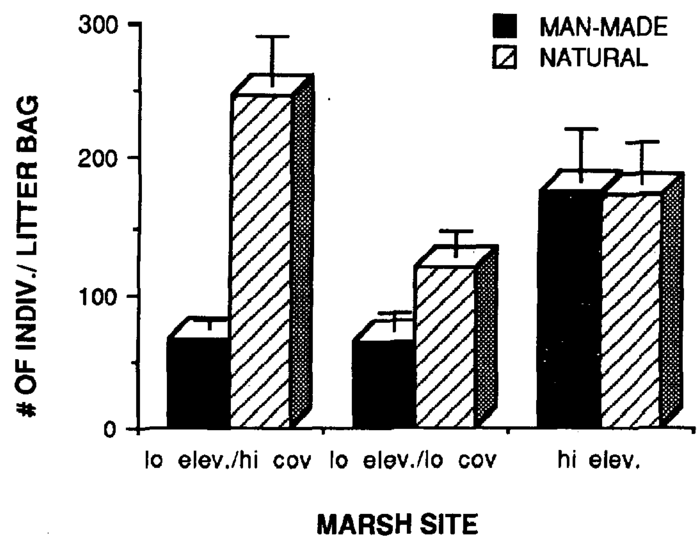
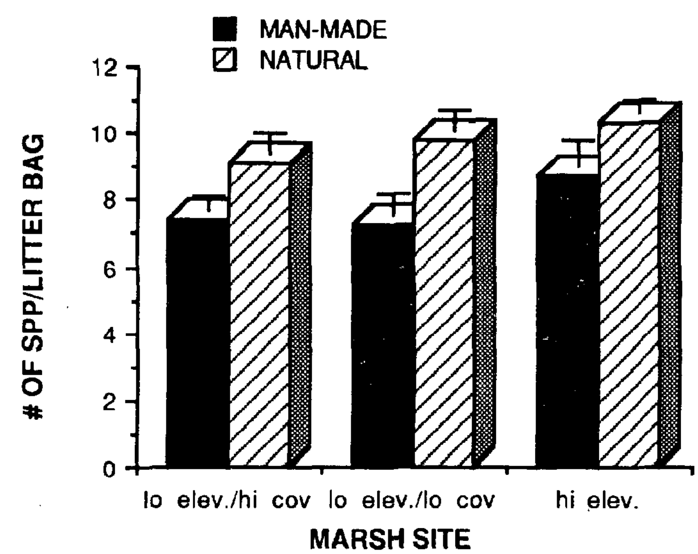


Figure III.9. Mean number of species per litterbag in areas of low elevation with high (80-100%) vs. low (0-20%) cover of cordgrass vs. high elevation with high cover.



Channel fishes and benthic invertebrates. The requirements for functional equivalency of the channel fish and invertebrates in the Biological Opinion (US FWS 1988) were that the constructed marshes provide 75% of the species and 75% of the individuals found in the reference marshes. An extensive sampling program was designed to understand how the constructed channel system has developed and to assess its functioning in relation to natural channels.

Channel organisms were sampled at three stations in the Connector Marshes and four in the natural (reference) marshes within SRWC (Table III.4). Fish and invertebrates were sampled in June 1989, October 1989, and January 1990. At each station, blocking nets were used to create a sampling area approximately 10 m long. A beach seine was repeatedly pulled through the sample area until the fish catch per unit effort declined. Individuals were identified, measured and released. Benthic invertebrates were sampled adjacent to each station with 9 cores using a clam gun (0.018 m^2 surface area) to a depth of 20 cm. Samples were preserved and retained for laboratory identification.

The fish community. In comparing the total fish caught over the three sampling dates, the natural marshes contained 15 species of fish. The Connector Marsh stations, when combined, contained all of these species and more. The average number of individuals caught at the connector marsh stations over the three sampling periods was 630, compared to an average catch of 610 fish at the reference marsh stations. This indicates that for year 5, with 3 sampling periods, the Connector Marsh has 100% of the species and similar numbers of individuals as are found in the reference marshes at SRWC.

The evaluation of functional equivalency is tempered by the occurrence of two exotic species in the Connector Marsh: yellowfin gobies (*Acanthogobius flavimanus*) and sailfin mollies (*Poecilia latipinna*). With the exception of one sailfin molly found in F/G Street marsh, these species were found exclusively in the constructed marshes. Generally, opportunistic nonnative species such as these are much more successful in colonizing disturbed habitats. Little is known about the specific impact that these species have on the native species. Yellowfin gobies are identified as voracious predators, and it may be assumed that they have some type of impact on the native community through predation. Likewise, sailfin mollies are very similar to killifish in their life history, and may have a competitive effect on this species. The effect of these species on the channel community should be investigated further.

The sediments and channel configuration of each station differed, providing a spectrum of habitat types within the wetlands complex (Table III.4). Because of this, a generalized comparison of the Connector Marsh stations with all of the reference marshes would be insufficient to evaluate the specific stations. Instead, a *similarity* index was used to compare the individual sites.

For each sampling station, the numbers of individuals sampled was totaled for each species over the three sampling dates. The relative abundance of each species was calculated for each station (Table III.5). From these relative abundances, an index of % similarity was calculated (Sim. = Σ[min. a,b], where a and b are relative abundances in the two samples being compared).

Table III.4. Description of channel stations at the Connector Marsh and other sites of the Sweetwater River Wetland Complex (SRWC).

Station	Description
Constructed channels (Connector Marsh, CM)	
Stn.2 South CM east channel	Coarse mud bottom. Steep bank on one side, gently sloping on the other. Max depth at low tide is approx. 0.75 m. Occasional deeper areas with 50 gal. barrels on the bottom.
Stn.3 South CM west channel	Soft mud, sloping banks on both sides. Completely intertidal (max depth at low tide is approx. 0.0 m).
Stn.7 North CM	Soft mud, with medium sloping banks on both sides. Max depth at low tide is approx. 1.0 m.
Natural channels	
Stn.1 Isla Flaca, Eastern tip	Sandy mud with gentle, slope on one side and steep, eroding bank on the other. Max depth at low tide is approx. 0.75 m. Sweetwater River flows diectly out this course.
Stn.4 Sweetwater Marsh, main channel	Soft mud/silt channel; some areas with dense clam shells (dead). Steep banks up to the marsh plain on both sides. Max depth at low tide is approx. 1.25 m.
Stn.5 E St. Marsh, main channel	Soft mud channel with varying depths. Generally steep banks on both sides to a marsh plain. Max depth at low tide is approx. 0.75 m.
Stn.6 F/G St. Marsh, main channel	Very soft mud/silt bottom. Low bank that is steep on one side and sloping on the other. Sluggish but regular tidal flushing. Max depth at low tide is approx. 0.75 m.

Comparisons of the fish communities within the Connector Marsh (Table III.5) indicate 72% similarity (of relative abundances) between the two stations in the west channel (South west and North west). Both western channel stations were dominated by topsmelt. These similarity comparisons also indicate that the fish community in the eastern channel (South east) is only about 20% similar to each of the other CM stations. The eastern channel is shallow, has a soft-bottom, and is dominated by arrow gobies. The eastern channel has changed considerably since construction of the Connector Marsh. Prior to 1984, it was the main tidal channel that directed flows.

Table III.5. Similarity among Connector Marsh stations. Comparisons are based on relative abundances of individuals for 3 seasonal samples in 1989-90. (Sim. = Σ[min. a,b], where a and b are relative abundances in the two samples being compared)

CM Stations	% Similarity
South west x North west	72.2%
South west x South east	20.4%
North west x South east	20.8%

to Paradise Creek. After dredging of the western channel, flows shifted to that deeper course, and sediments accreted in the east channel. It is understandable that the eastern channel has a different fish community from the deeper channel that was recently constructed.

The comparison of CM sampling stations indicates that there are at least two habitat types represented at the Connector Marsh: 1) a mudflat/channel habitat (east channel) that is dominated by arrow gobies; and 2) deeper, subtidal channel habitats (west channels of North and South CM) that are dominated by topsmelt. Comparing species lists for the Connector Marsh and reference channels did not reveal this difference.

The three Connector Marsh stations were then compared to those for the natural marshes (Table III.6). Data from the deeper, western channel of both North CM and South CM are most like those from the channel at Sweetwater Marsh (56-64% similar). The Sweetwater Marsh channel has a high proportion of topsmelt. In contrast, the shallow, soft-bottom habitat of the eastern channel of CM is most similar to Isla Flaca. Both of these stations are distinguished from the others by their large, adjacent intertidal mudflats, and both are dominated by arrow gobies.

Conclusions regarding fish communities. The *preliminary* conclusion, based on 3 seasonal samples, is that the US FWS criteria for fish species and abundance have been met, since all of the native fish species have been shown to occur in the constructed marsh channels and since densities of fishes are very similar to reference channels.

Two functional differences need to be explored further. First, non-native species have colonized the constructed marshes, and their influence on native populations is unmeasured. Second, the fish community (relative abundances) of the new channel of the constructed marsh does not closely resemble that of natural channels. The highest similarity was 71%, comparing the accreting channel along the eastern side of South CM with the Isla Flaca channel, which receives freshwater inflows from Sweetwater River. The most obvious difference between reference channel fish communities and CM channel communities is that several species tend to share dominance in the reference channels, while CM channels are more heavily dominated by a single species. A more thorough evaluation of this pattern is needed.

Table III.6. Similarity of fish sampled in natural and constructed wetland channel stations. Data are for three seasonal samples in 1989-90. Similarities are based on relative abundances of species, as in the previous table.

CM Channel	Sweet-water	E St.	F/G St.	Isla Flaca
North west	**56%**	41%	37%	34%
South west	**64%**	47%	27%	39%
South east	56%	49%	15%	**71%**

Benthic invertebrate community. Benthic invertebrates were sampled adjacent to each fish sampling station with 9 cores using a clam gun (0.018 m^2 surface area) to a depth of 20 cm. An additional station was established in Vener Pond where shorebirds are known to feed. Invertebrate samples were preserved and retained for laboratory identification.

Sampling of the benthic invertebrate community of the constructed marsh is not complete. Laboratory examinations and counts have been done only for the June and October 1989 sampling periods.

The following summary of findings is thus preliminary.

In all, 27 taxa were captured and identified throughout the wetland complex; of these, 14 were found in numbers of fewer than 5 in the combined June and October samples (Table III.7). Despite the use of 9 cores per sampling station (a total area of 0.16 m^2), many stations had very low densities. The results obtained to date show high spatial and temporal heterogeneity in the sand and mud cores, with several station samples having few animals, and one station (Vener Pond in June) containing 173 individuals of a small snail. Combining both sample dates, totals encountered ranged from 20 to 276 animals.

The total number of individuals sampled was 683; standardizing the numbers to area sampled indicates that samples from the constructed channel had 54% as many individuals as the natural benthic sampling stations.

Of the 27 taxa, 11 were found in both constructed and natural channel stations; there were 20 taxa in the natural channels and 18 in the CM sampling stations. Assuming no effect of greater sampling area (an assumption that may not be met since there were 4 natural stations and 3 constructed stations), the similarity of species lists is 58% (using the 2w/[a+b] similarity index). Comparing the 11 taxa of the constructed channel to the 20 found in natural channels suggests that only 55% of the naturally occurring taxa have established in the Connector Marsh channels by the October sampling date. These are preliminary calculations, and the similarities will most likely change once the full-year data set is analyzed.

Preliminary summary of benthic invertebrates. Of the two seasonal samples for which animals have been identified and counted, the constructed channels have 55% of the species and 54% of the densities of individuals as in the natural benthic sampling sites. These calculations are clearly preliminary and are not conclusions of the study.

The bird community. The objectives of marsh restoration include provision of foraging habitat for endangered birds. Understanding how the entire bird community uses the restored marsh is important to the overall functional assessment. Not only can we learn what functions the restored marsh provides, we can also explore relationships among species and species groups to understand why certain uses occur or fail to develop.

Extensive bird surveys were undertaken (Ashfield and Kus, unpub. data). Detailed analyses and results will be provided in an M.S. thesis (Ashfield, in prep.). The following is an initial evaluation of the data from the Connector Marsh, the Paradise Creek, and the Bay Shoreline. Sampling took place twice per month for 13 months from March 1989 through March 1990 (except once per month in November 1989 and January 1990), for a total of 24 low-tide surveys. We report on the information for low-tide surveys only (high-tide uses were also recorded). We include only the waterbirds (excluding gulls).

Two sampling sites were selected for comparison with the Connector Marsh (CM): Paradise Creek (PC) and the Bay Shoreline (BS), which is just west of Gunpowder Point. The rationale for comparing CM to PC is that PC habitat was damaged and replacement of functional losses there was part of the objective of constructing CM. The reason for comparing CM to BS is that CM has a larger proportion of intertidal flat than PC.

Table III.7: Channel invertebrates found at the Sweetwater River Wetlands Complex, June plus October 1989. Data are numbers per sampling station (number/0.162 m^2).

Taxa	F/G St.	E St.	SW proper	Vener Pond	Isla Flaca	S.isl. west	North Isls.	S.isl. east	Total
Bivalves									
Macoma nasuta									
Tagelus californianus	1	0	2	0	0	1	2	4	10
Protothaca staminea	4	16	37	0	2	18	10	25	112
Musculista senhousei	1	1	3	0	5	3	2	2	17
Gastropods	2	0	0	0	0	0	0	0	4
Assiminea californica									
Cerithidea californica	0	0	0	0	0	6	0	0	6
Acteocina inculata	9	1	13	16	1	30	3	13	86
Bulla gouldiana	16	0	0	173	11	0	0	0	200
Nassarius sp.	0	0	0	0	0	0	0	1	1
Crustacea	0	0	0	0	1	0	0	0	1
Hemigrapsus oregonsis									
Amphipoda	1	1	4	0	0	0	0	1	7
Orchestia traskiana									
Anthozoa	0	0	0	0	0	9	0	0	9
Ceriantharia									
Polychaetes	1	0	0	0	0	0	0	0	1
Capitellidae									
Unknown capitellids									
Cirratulidae	0	0	0	0	0	2	2	1	5
Cirratulus cirratus									
Tharyx sp.	0	0	1	0	0	0	0	0	1
Eunicidae	2	0	0	0	0	1	0	0	3
Eunice valens									
Glyceridae	0	1	0	0	0	0	0	0	1
Nereidae	0	0	1	0	0	0	0	0	1
Nereis sp.									
Orbiniidae	0	0	0	0	4	6	1	5	16
Scoloplos armiger									
Phyllodocidae	0	0	15	0	10	0	0	0	25
Eteone sp.	0	0	0	0	2	1	0	0	3
Spionidae	0	0	0	0	0	0	1	0	1
Polydora nuchalis									
Polydora cornuta	11	0	0	26	0	3	0	0	40
Polydora sp.	2	0	0	2	0	12	0	6	22
Streblospio benedicti	26	0	0	59	0	12	4	10	111
Fly larvae	0	0	0	0	1	0	0	1	2
Chordata	0	0	0	0	0	0	0	2	2
Ascidacea									
Diplostoma sp.									
	1	0	0	0	0	0	0	0	1
Totals	77	20	76	276	37	104	25	71	688

The three census sites are not identical in size or physiography. The Paradise Creek site was larger but, because it is dominated by salt marsh habitat, it had the smallest area of intertidal flats. Field estimates of the areas sampled at CM and BS were very similar at 6-7 ha. Of these, the BS site was entirely intertidal flat habitat, while CM was a combination of flats, channel water, and marsh.

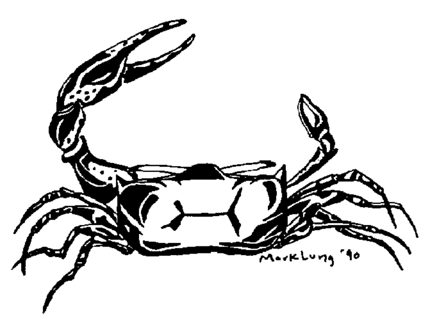
Comparisons of CM with both PC and BS. Overall, the number of water-associated bird sightings was lowest in PC (less than 1/3 that at CM), despite the fact that this census area was larger than CM and BS (Table III.8). In the low-tide censuses, excluding gulls, PC had 23 species of waterbirds, CM had 24, and BS had 39. Using these data, PC and CM had about the same species richness, but attracted fewer species than BS. Note that all sites would have longer species lists if gulls, high-tide surveys, and land birds were included.

Neither the lower numbers of sightings nor fewer species seen at PC can be attributed to sampling area, since PC was the largest site; hence, the finding of fewer sightings at PC than at both CM and BS is real. The extensive cover of salt marsh vegetation at Paradise Creek and its single large channel (rather than a highly dissected tidal creek system) is not optimal habitat for large numbers of water-associated birds. PC does, however, support dowitchers, willets, and western sandpipers in fair abundance.

The largest number of sightings was at BS (nearly 3x as many as at CM and over 9x as many as at PC). With its high species richness and abundance of waterbirds, BS is clearly the most-used waterbird habitat of the three comparison sites. CM is intermediate in waterbird support functions, between PC and BS.

The three sites differed in the number of species encountered, in sightings of western sandpipers, and in the area of intertidal flats. Because western sandpipers are so abundant at BS, their numbers mask the importance of other species. The largest total for dowitcher sightings was at BS, as were totals for willets and marbled godwits.

CM shares 18 species with PC and 18 with BS (Table III.8). The data on relative numbers of sightings show moderate similarity for the top ten species, which comprised 80-90% of all sightings. The similarity between CM and PC (77% for species lists and 62% for sightings, Table III.8) was greater than the similarity between CM and BS (57% for species lists and 53% for sightings). Thus, despite the fact that CM and BS are more alike in having large areas of intertidal flats, CM and PC appear more similar in their bird communities. As an intertidal flat, CM functions less well than BS, both from the standpoint of numbers of species and numbers of sightings. As a channel-marsh system, CM and PC are moderately similar in bird communities, with substantially more numbers of water-associated birds seen at CM than at PC.



Fiddler crab
(Uca crenulata)

Table III.8. Sightings of water-associated birds at three wetlands of the SRWC. Data are for 13 months in 1989-90. Data are for low-tide and water-associated species excepting gulls. The similarity of species lists was calculated as 2w/a+b, where w=no. of species in common, a=no. of species at PC and b=no. of species at CM. Relative data are provided only for species with at least 5% of the sightings in at least one of the three sites. Similarity of relative sightings was calculated as Σ(mininum a,b), where a=rel. sightings at PC and b = rel. sightings at CM, summed for all species, not just the ones listed here).

Summary data	PC	CM	BS
Total sightings relative to x	0.31x	x	2.88x
Number of species sighted	23	24	39
Relative number of sightings (% of site total)			
Western sandpiper	15.5	38.9	40.6
Dowitcher spp.	25.0	29.1	11.7
Willet	20.0	3.5	7.2
Marbled godwit	4.9	2.1	8.5
Least sandpiper	2.1	9.7	0.7
Dunlin	5.7	4.8	1.7
Killdeer	4.3	6.5	0.1
Red knot	0	0	8.7
American coot	0	7.0	0
Bufflehead	6.6	0.1	0
Sum for above 10 species (%)	84.1	91.7	79.2

Site comparisons (all species)	CM with PC	CM with BS
Number of species in common	18	18
Similarity of species lists	76.6%	57.1%
Similarity of rel. numbers of sightings.	62.5%	52.7%

Conclusions from the bird census data. The avifauna of SRWC is abundant and species rich. The Connector Marsh provides habitat that is somewhat like that of the marsh-dominated Paradise Creek site and somewhat like that of the Bay Shoreline. The relative numbers of bird sightings at CM are more similar to those of PC than the Bay Shoreline, indicating that the CM site can support many of the marsh birds (e.g., dowitchers, willets) but that it lacks some of the qualities of the best mudflat site of SRWC. Overall, there were more sightings at CM than PC, but far fewer than at BS.

Overall evaluation of functional equivalency. It is not easy to create cordgrass ecosystems that are functionally equivalent to natural ones. In a recent attempt to simplify the complex data comparing the two types of systems at SRWC, we summarized 11 data sets that were gathered when the constructed marsh was 4-5 years old (Zedler and Langis in press). The resulting "index" (Table III.9) indicates that the constructed marsh was less than 60% functionally equivalent to the natural reference wetland (Paradise Creek Marsh).

Table III.9. Functional equivalency of the constructed and natural cordgrass marshes comparing soils, nutrients, plants, and epibenthos. Soil and plant data are from Langis et al. (in press) except for plant heights (unpub. PERL data). Epibenthic invertebrate data are from Rutherford (1989).

Data set	%
Organic matter content	51
Sediment nitrogen (inorganic N)	45
Sediment nitrogen (TKN)	52
Pore-water nitrogen (inorganic N)	17
Nitrogen fixation (surface cm)	51
N fixation (rhizosphere)	110
Biomass of vascular plants	42
Foliar nitrogen concentration	84
Height of vascular plants	65
Epibenthic invertebrate numbers	36
Epibenthic invertebrate species lists	78
Average of comparisons =	**~57%**
95% confidence limits =	40-74%

Inclusion of data on fishes, channel benthos, and birds failed to increase the overall similarity. Although fish species lists were 100% similar for the two sites, the bird for which the marsh was designed (light-footed clapper rail) was not yet using the constructed marsh.

A major difference between the constructed and natural sites is easily seen during high tides. Tall cordgrass in the natural marsh extends above the water, providing a refuge for terrestrial arthropods and cover for birds hiding among the plants. In contrast, cordgrass in the constructed marsh is completely inundated, and the habitat islands disappear from sight.

The shortcomings of the constructed marsh were not anticipated; on the contrary, the site benefitted from extensive planning and biological advice from several agencies. Detailed assessments, such as the work reported here, had never been done in our cordgrass marshes, and no one knew what site characteristics needed to be measured to predict what would limit ecosystem development.

The quantitative comparisons of constructed and natural cordgrass marshes (Langis et al. in press) revealed significant differences in substrate characteristics that help to explain why transplanted cordgrass is growing poorly. The sandier soils (probably from alluvial outwash) have little organic matter and little nitrogen. The low organic matter content limits nitrogen fixation and nutrient recycling by microbes. Thus, cordgrass growth is limited. Because levels of these two causal factors did not increase during the study, we reserve judgment on how long it will take for functional equivalency to reach acceptable levels. (Funds have been requested to continue comparisons through age 9.)

Our understanding of the importance of soil organic matter and nitrogen suggests corrective measures for future restoration sites, and experiments are now underway to test the ability of a variety of soil amendments to accelerate the development of constructed cordgrass marshes. As the science of habitat restoration advances, it should be possible to achieve greater than 60% functional equivalency. At the same time, as scientists continue to understand the details of how natural wetland ecosystems function, expectations for restoration sites will also rise and the attributes that are assessed will need to expand accordingly.

IV. Sampling methods and comparative data from natural wetlands

Very few restoration or mitigation projects have included adequate monitoring programs, either in the extent of information gathered or the length of time sampling has continued. In a review of Section 404 permits (Clean Water Act) in Washington State, only 31% of the permits were found to require monitoring; those that did were typically of short duration (usually 1 year), and only required data on vegetation coverage (cf. Josselyn et al. 1989). Likewise, the California Coastal Commission's Wetlands Task Force (Ray et al. 1986) found that the effectiveness of mitigation efforts has not been documented--56% of permits reviewed required some kind of monitoring, but efforts ranged from simple visual observations to water quality changes, with results that were not comparable. In many cases, results were not available and it was suspected that monitoring was not performed.

There are three basic recommendations for an assessment program:

First, use standard methods and compare data with existing monitoring programs. The methods proposed for monitoring are taken from published literature and/or have been used to monitor other southern California coastal wetlands (Tijuana Estuary, San Diego River Marsh, Los Penasquitos Lagoon, Mugu Lagoon). The most important considerations are the type of gear used to collect organisms (e.g., mesh size of fish seines), time of sampling (e.g., soil salinity varies seasonally), size of sample unit (e.g., data on frequency of occurrence of plant species vary with quadrat size), and location of sampling station (small changes in topography affect species composition). Persons who supervise and carry out sampling should interact (e.g., exchange reports) with personnel from the Pacific Estuarine Research Laboratory who monitor San Diego County wetlands in order to access the most recent reference data sets.

Second, use a hierarchical approach, insuring broad coverage with the most general descriptors of ecosystem condition (e.g., areas covered by tides) and selected, individual sampling sites for additional characteristics (e.g., plankton).

Third, obtain aerial photos annually and identify changes from year to year. The US Army Corps of Engineers photographs the coastline each year and has air photos available for purchase. Determine the tidal condition on the date and time that the photo was taken from tide tables. Photos at maximum high tide will indicate tidal coverage but obscure vegetational changes, while photos at low tide will show the development of tidal creeks in areas formerly dominated by plants. Both will be useful for interpreting changes in the overall ecosystem. Use photos to identify locations where tidal creeks are developing, to track the expansion of vegetation onto newly graded tidal flats, to document changes in the linkages between the estuary and the surrounding landscape, and to quantify the portion of the adjacent landscape that is developed. Aerial photo analyses should extend to areas beyond the wetland to include potential enhancement sites.

Recommendations for measuring ecosystem conditions and functions may exclude some measures that are common in other regions because their measurement is too difficult or the results are not meaningful in this region. The prime example is net annual primary productivity of vascular plants (NAPP) and algae. Both are highly variable, the former on an annual basis, the latter on a weekly basis. In addition, NAPP is grossly underestimated but not to the same degree for each species in the region (Onuf et al. 1978). Thus, it is nearly impossible to obtain data that can be compared from site to site.

Table IV.1. Ecosystem attributes to be considered in assessing how well constructed wetlands replace the functions of natural wetlands:

Attribute and Measures	Reason for Analyses
Hydrology*	
Current velocity and distance from tidal inlet	Tidal circulation
Water levels at various tidal cycles	Tidal lags, inundation regime
Salinity of water and soil*	Relation with streamflow
Topography	
Elevation*, slope	Erosion, accretion
Soils	
Texture	Drainage, resilience of soil
Organic matter*	Nutrients, resilience of soil
Toxic substances	Biological accumulation
Redox	Indicates drainage, organic matter, anaerobiosis
Sulfides and pH	Potential for acid sulfate soil formation
Nutrient dynamics	
Nitrogen fixation rates	Availability to producers
Inorganic nitrogen in sediments and pore water*	Potentially limiting nutrient
Denitrification rates	Nitrogen recycling
Organic matter decomposition	Nutrient mineralization
Nitrogen mineralization rates	
Algae	
Cover by dominant type*	Food for invertebrates Potential for nuisance blooms
Vascular plants	
Total stem length (m/m^2) of cordgrass	Estimates standing crop
Cover of vascular plants*	Shifts in dominance
Density of rare annual plants	Population persistence and growth
Consumers	
Decomposers and shredders	Food chain support
Aquatic insects	Indicators of water quality, food chain support
Terrestrial insects, especially pollinators and predatory insects	Food chain support; control of herbivorous insects, pollination
Fishes* and invertebrates*	Food chain support
Birds*	Food chain support
Reptiles and amphibians	Food chain support
Mammals	Food chain support

*Highest priority components to assess

1. Hydrologic functions

The most important forcing function of a coastal wetland is its hydrology, and readers are referred to hydrologists to understand the hydrodynamics of coastal wetlands. In almost every management planning effort, the bulk of the effort goes into the characterization of the hydrology of the system and use of models to predict changes in the depth and circulation of the system under different management (e.g., dredging) regimes. This manual gives only the briefest introduction to hydrologic functions.

Objectives. The objectives of a hydrologic survey are often to determine the system's tidal prism and the size of tidal prism needed for self-maintenance of the ocean inlet, to characterize tidal flushing, and to understand the influence of tidal circulation on the ecosystem. Three hydrologic features are of special ecological interest--inundation patterns, salinity regimes, and water column stratification. Understanding the flooding and drainage patterns of tidal channels is also extremely helpful in selecting sampling times for fishes and other channel organisms. Knowing how the system deviates from predicted tide levels on standard tables helps one arrive at the sampling station when the water levels are appropriate for the type of data being collected. Maximum and minimum water levels within a tidal system can differ by 2 hours or more.

Inundation regime. Remote data loggers are used to measure depth and duration of tidal inundation (to estimate tidal lags and tidal amplitude damping) for selected tidal cycles prior to wetland alteration and following construction of wetland habitats. Measuring tidal amplitudes and lag times will show how much tidal circulation was improved by excavation of intertidal habitat. Placement of tidal staffs (simple, graduated measuring sticks) facilitates location of data loggers at comparable elevations. Measurements are made in both man-made and reference wetlands simultaneously to determine if the constructed system behaves similar to a natural wetland. Measurements at increasing distances from the tidal inlet help to reveal any obstructions or bottlenecks in the main channels. The identification of a sill or hardpan that restricts tidal flows will help determine whether there is potential for increased tidal prism and whether such an obstruction needs to be removed. Finally, measurements before and after the construction of new intertidal habitats should show the effects of the newly created marsh on total-system hydrology.

To characterize tidal flushing, the tidal heights are plotted with time (24-hr cycle) for each location. On the coast, the minimum tidal amplitude occurs in spring and fall, and the maximum in summer and winter; lowest tide levels occur in daytime in winter and at night in summer. Tidal conditions are modified within the estuary. Reduced amplitudes and more delayed peaks occur at increasing distances from the ocean inlet. Increased amplitudes and shorter lag times are anticipated following dredging or grading operations that increase tidal prisms, and such measurements would indicate improved tidal flushing.

Elevation. The hydrologic conditions of intertidal sites are determined by measuring elevations relative to the National Geodetic Vertical Datum, i.e., the 1929 mean sea level, which is an average of data for the preceding 19 years. Distributions of salt marsh plants are often referenced to this standard datum; however, it must be remembered that the inundation regimes of a specific elevation may differ for various locations within an estuary. Because tidal maxima and minima are damped at the inland extent of tidal creeks, and because peaks lag behind conditions at the ocean inlet, the inundation regime for an elevation of 0 ft or m NGVD at the mouth may differ significantly from that at the most inland edge of the estuary. In addition, microtopographic features of the intertidal zone may impede inundation or drainage, such that one site at 1 ft NGVD may be well drained, while another impounds high tide waters. Thus, elevation is a general indicator of inundation regimes, not a precise measure of habitat conditions. Elevation should be measured as a general descriptor, but one should not assume that the same absolute elevation will have the same environmental conditions.

High precision is needed in elevation surveys. Salt marsh vegetation is extremely sensitive to slight differences in tidal inundation, and plants that thrive at one elevation may yield to another species if the topography is 10 cm (4 inches) too high or too low. For mapping of wetland habitats and for marking sites for vegetation transplantation, 30-cm (1-foot) contours are the coarsest intervals that are useful.

Elevations are measured relative to benchmarks near the estuary. Professional surveyors may be needed to establish benchmarks near study sites, if these are not present. An automatic level (e.g., Wild Instruments) and calibrated stadia rod are used to measure and/or mark additional elevations at the site. Elevations of all systematic sampling stations are determined in this manner. While reference data are nearly always in feet and inches in US publications, the metric equivalents should also be reported for international comparisons.

Water Column Stratification. Impaired tidal flushing can also be detected through measurements of water temperature and salinity. In sluggish channels, water columns become stratified, with surface and bottom water differing in either temperature or salinity. In winter, following rainfall and streamflow into the estuary, fresher (and possibly warmer) water may float over the more saline/cooler seawater. In late summer, the pattern of stratification may reverse, with warmer hypersaline water overlying seawater.

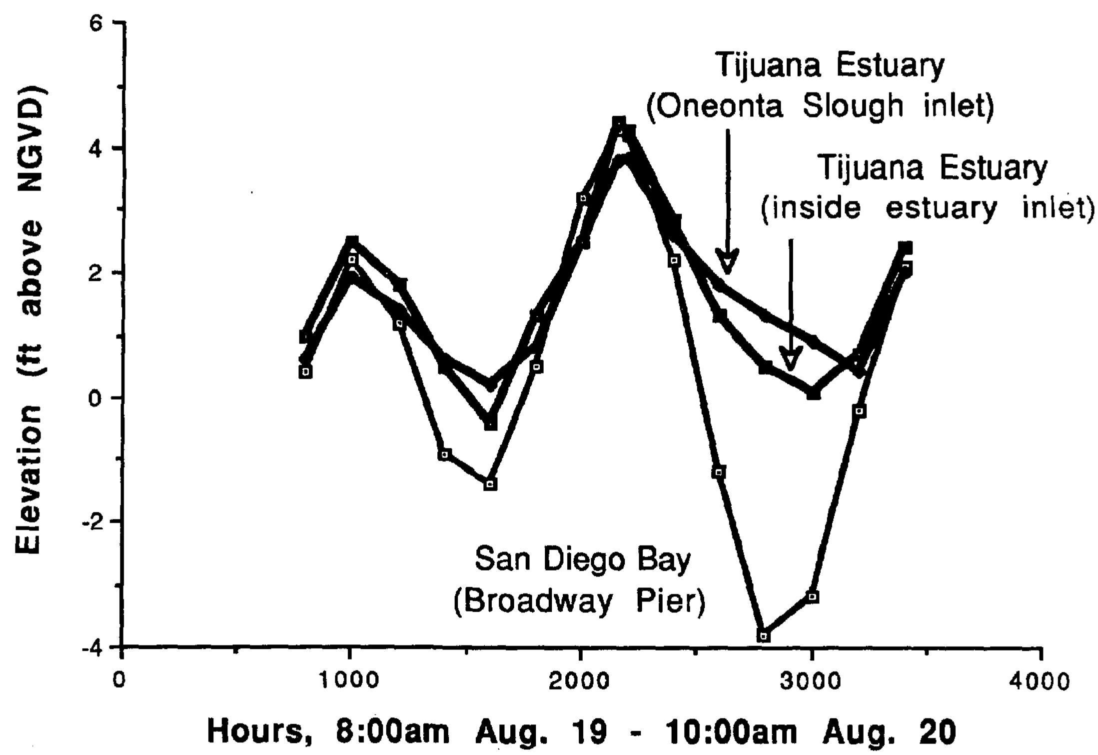
To characterize water column stratification, water salinities and temperatures are taken monthly at selected sampling stations. Temperature is first taken at the surface and at the bottom; if there are differences, additional sampling at 10-cm vertical intervals should be done to determine where the thermocline exists. Since estuarine water temperatures vary with the tidal condition, the time of day, the storm condition, and the season, these measurements are more useful for determining whether the water column is stratified, than for characterizing an "average" water temperature. To obtain average water temperatures, we recommend using continuous sampling with a data logger to record conditions during both spring and neap tides, over 24-hour periods. Such detailed data would be needed for scientific studies and modeling of water circulation.

Additional indicators of poor tidal flushing include phytoplankton blooms (pea-soup green water) and dense mats of macroalgae. Macroalgal cover can be estimated at the water salinity stations, distinguishing epibenthic from floating algae (see water quality methods).

Reference data. Williams and Swanson (1987) compared tidal flushing data from several Tijuana Estuary stations with data from a nearby tide recording station (San Diego Bay at Broadway Pier). They found differences in tidal amplitudes and in the times of minimum and maximum water levels. Data from three stations are replotted in Figure 1.1. The greatest differences between stations within Tijuana Estuary and the Bay are in the amplitude between higher high and lower low water and the lag in outflow following the higher water. The large volume of water flowing into the estuary at higher high tide was held for a longer time than the inflows of the lower high tide. Flows out of Tijuana Estuary were constrained by the shallow ocean inlet. It was also clear that flows out of Oneonta Slough were constrained by the narrowness of that channel's inlet, which was due to overwash fans (ibid.). These overwash constrictions to tidal flushing were removed by dredging in 1987 and an improvement in flow to Oneonta Slough was documented by resurveys of channel cross-sections (Florsheim and Williams 1990).

Additional hydrographs are presented in Williams and Swanson (1987) to show other lags within Oneonta Slough in 1986. About 1.5 km (5,000 ft) inland along Oneonta Slough, higher high water lagged 2 hours behind that at its inlet and lower low water was a foot higher, indicating that inflows and outflows were both constrained upstream along the tidal channel.

Figure 1.1. Hydrographs for two locations at Tijuana Estuary compared to unconstrained tidal flows at San Diego Bay. Data redrawn from Williams and Swanson (1987).



2. Water quality

Objectives. Water quality measurements are needed to document problems, such as sewage spills, and to predict biological impacts, such as fish kills. The most thorough water quality investigations, i.e., those including tests of heavy metal concentrations and presence of organic toxins, will help to predict the potential and risks for invertebrates, fishes, and birds. In southern California, the greatest risks to marine animals develop when the mouth of an estuary or lagoon closes to tidal flushing. Initially, water temperature rises, so that saturation concentrations for dissolved oxygen decrease. Microbial activity increases, along with metabolism rates of all animals and plants; thus, overall oxygen consumption increases. While algae may become more productive, high biomass will accumulate in the surface water, shading plants in the channel and creek bottoms, so that less oxygen production will occur in the habitats occupied by benthos. Later, decay of algal cells and fronds by decomposers will increase oxygen demand, and bottom waters will become hypoxic (low in oxygen). In the absence of rainfall, evaporation will concentrate the saline water trapped in the estuary, and channels and creeks will become hypersaline. Thus, with closure alone, substantial changes in water quality will develop. The changes in Los Peñasquitos Lagoon, during the 1979 closure, illustrate these patterns (Figure 2.1-2.3). Comparative data are shown for sample dates before and after closure, as well as inside and outside the inlet.

Much higher salinities were documented at Los Peñasquitos Lagoon in the 1950's, when the lagoon remained closed to tidal flushing for several consecutive years. Channel waters exceeded 60 ppt in 1959 (Carpelan 1969). At Tijuana Estuary, during a long nontidal period, channel water salinity reached 60 ppt 7 months after closure. Estuary-wide die-offs of invertebrates (e.g., crabs and hornsnails) were noted.

More severe changes in water quality develop when inlet closure is combined with a sewage spill or rainfall and runoff event. Over the past several years, Los Peñasquitos Lagoon has closed to tidal flushing during the warm summer and fall months. On two occasions, major freshwater inflows and salinity-dilution events occurred. The first was a major sewage spill; the second, an early rainfall. Without tidal flushing, the non-saline waters were impounded, and extreme salinity dilution occurred, followed by major fish and invertebrate kills.

Additional data are provided by Nordby and Zedler (in press), including evidence from Tijuana Estuary and Los Peñasquitos Lagoon that lowered water salinity reduces the species richness of both fishes and benthic invertebrates.

What to measure. Several factors such as nutrient concentration, light attenuation, dissolved oxygen, salinity, and temperature are recorded to assess water quality. The aquatic organisms, i.e., zooplankton and algae, also help to characterize water quality. There are limited data on planktonic algae, but no reference data for zooplankton. Sampling before and after restoration measures take place is important to determine the changes in ecosystem condition.

Permanent sampling stations are chosen in a stratified random manner, to assure similar representation for all aquatic habitat types. Habitat types may include deep channels, tidal creeks, deep saline ponds, brackish and fresh sections of the incoming creeks, and freshwater ponds. Stations should be sampled biweekly, or at least monthly, to detect and account for seasonal as well as yearly patterns.

Figure 2.1. Changes in water temperature at Los Peñasquitos Lagoon, before and after closure of the inlet on May 12, 1979. The "Stagnant Arm" parallels the barrier dune and has reduced tidal influence year round; the sampling station under the railroad (RR) bridge is the deepest site in the lagoon. Data are single measurements from the water surface (PERL, unpubl.).

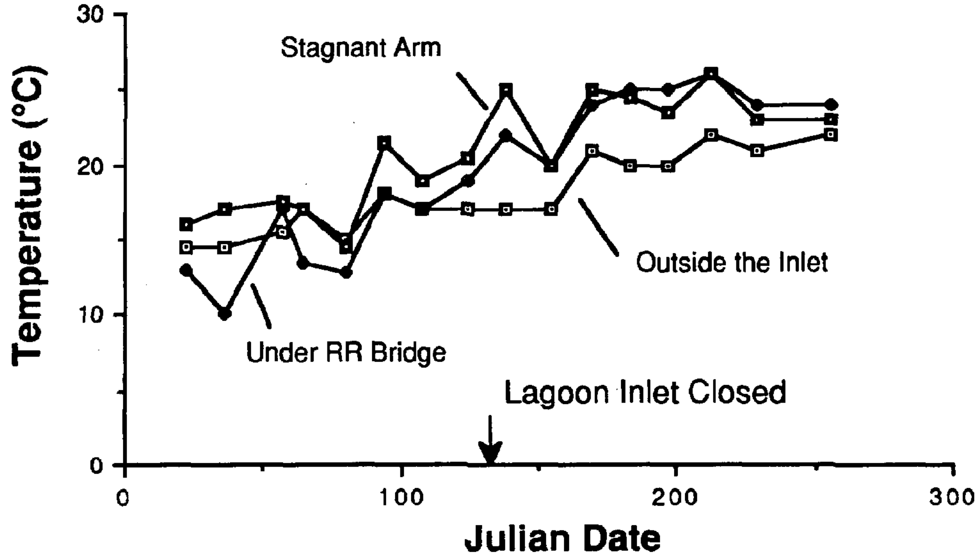
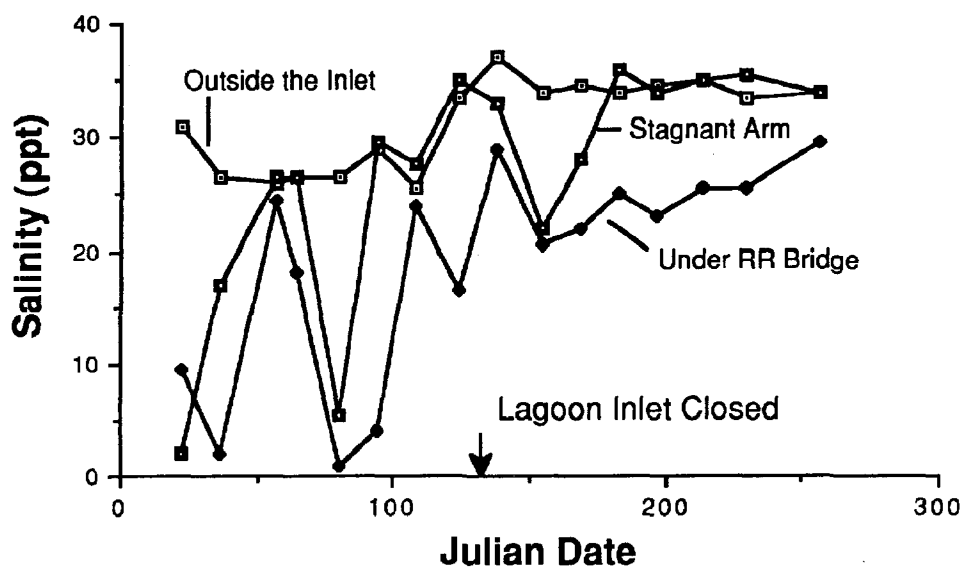


Figure 2.2. Reference data for lagoon water salinity at the surface, comparing sites before and after inlet closure, as in Fig. 2.1.



Water temperature and dissolved oxygen are measured using a dissolved oxygen-temperature meter (e.g., Yellowsprings Instrument Model YSI 51B).

Water salinity is measured to the nearest part per thousand using an American Optical (or Reichert) salinity refractometer. Alternatively, a YSI Salinity Meter and probe can be used for ease in sampling vertical salinity profiles. Both the refractometer and salinity meter read salinity in parts per thousand.

Light attenuation is measured using a submersible light meter (e.g., Li-Cor), sampling at the surface and at the bottom, with the depth between measurements recorded. Several different units can be used; the measurement of interest is % attenuation. The extinction coefficient is $k = [\log I_0 - \log I_z]/z$, where I_0 is the amount of light at the water surface and I_z is the amount of light at depth z. A simpler measure, for deeper water bodies is to lower a Secchi disk (a 20-cm diameter disk) from the shady side of a boat or pier and determine the depth at which it is no longer visible.

Nutrients are collected in water samples using the same techniques as for phytoplankton. Samples can be frozen and analysed for nutrient concentration at a later time, but more reliable measurements are obtained on fresh samples that have been kept on ice between the site and the lab. Wet chemistry techniques for nutrient analysis are time consuming and routine analyses are best conducted on an autoanalyzer (e.g., Technicon Instruments Inc.), which automates the process.

Algae. In aquatic ecosystems, algae are the base of the food chain. While measurements of algal populations are not very good estimators of primary productivity, they are useful indicators of eutrophication and tidal flushing (cf. Figure 2.3). When phytoplankton accumulate to bloom proportion, anaerobic conditions can develop at the channel bottom during the night. In tidal channels, the highest algal biomass would be measurable at low tide at the end of a neap tide series, when channels would not have been greatly diluted by seawater.

In each of the aquatic habitat types, water samples for phytoplankton analysis can be collected in a simple plexiglass tube of 2-cm internal diameter, which is the length of the water column being sampled (e.g., 1 m). This tube is lowered vertically into the water column to ensure sampling from all strata, then sealed at the top and drained into the sample bottle. This sampling can be concurrent with the quarterly zooplankton sampling, utilizing the same stations.

Phytoplankton biomass is estimated as chlorophyll *a* concentration using either the fluorometric technique or extraction and measurement of absorbancy using a spectrophotometer. The data in Figure 2.4 were obtained with the latter method, extracting the pigment in 90% acetone, measuring absorbancy at 663, 645, and 630 nm (subtracting absorbancy at 750 nm from each reading), and applying the formula, Chl *a* $= 11.64Abs_{663} - 2.16Abs_{645} + 0.1Abs_{630}$ (Strickland and Parsons 1972). Results (mg pigment per liter of acetone), divided by liters of seawater filtered, will yield micrograms of pigment per liter of seawater (= milligrams per cubic meter).

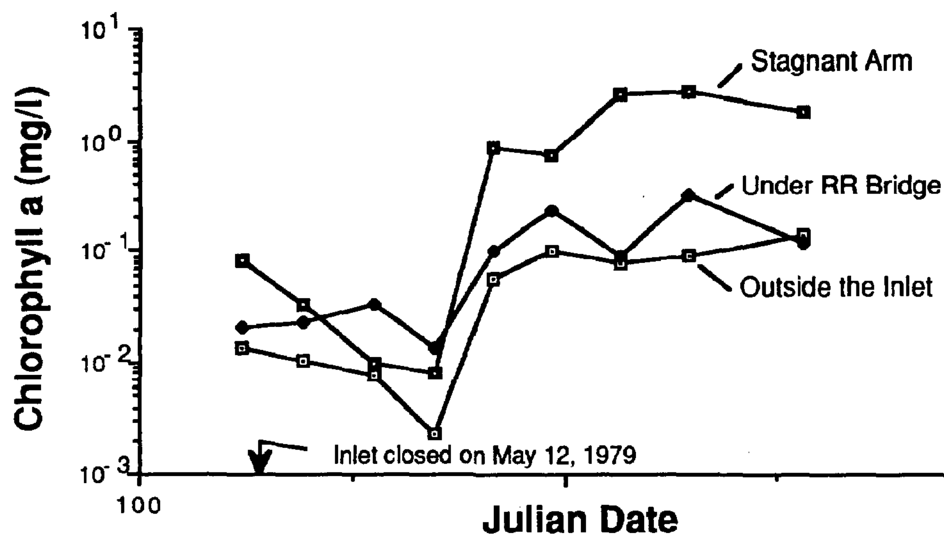
At present, there are no exhaustive data sets for coastal wetland channels. However, the San Diego Regional Water Quality Control Board has comparative data sets for several lagoon sampling stations over a limited time period. Experimental studies (Fong 1986, Fong et al. 1987) have addressed the role of nutrient additions to coastal water bodies.

Visual estimates of the percent of the water surface covered by macroalgae should be made at the same stations, and the genus noted (usually members of the Chlorophyta, either *Enteromorpha* or *Ulva*). Permanent plots are marked for cover estimation, using a rectangular shape to reduce variance, and a size that is appropriate for the habitat type (smaller for tidal creeks than main channels).

Zooplankton. Sampling for zooplankton can be done quarterly at the same stations as the fish and invertebrates, although prior to or on a different day, as the seining often resuspends benthic particles, obscuring the samples. Experience has shown that sampling for plankton is most efficiently accomplished by boat as the water is not disturbed as much as when walking. This is true for the zooplankton, algae, and physical/chemical sampling, and all can be accomplished at the same time. As these communities should be sampled at high tide, all of the habitat types can be included.

Plankton nets with a mesh size of 35 microns are appropriate for collection of zooplankton samples. As most of the habitats are relatively shallow, oblique tows from channel or lagoon bottom to the surface are made behind a boat that is travelling parallel to the shoreline. Samples should be fixed in the field in formalin, and quantified microscopically using Sedgwick-Rafter counting chambers. Zooplankton densities and community composition can be assessed spatially and temporally. Sampling of plankton should be done seasonally under the same tidal condition (e.g., end of a neap tide series), in order to reduce the effect of seawater dilution.

Figure 2.3. Comparisons of phytoplankton (as relative amounts of chlorophyll *a*) at Los Peñasquitos Lagoon. One sample was taken prior to inlet closure. Phytoplankton concentrations built up in the poorly circulated "Stagnant Arm" following closure. Note the log scale. Due to questionable calibration of the spectrophotometer, absolute values for concentrations are not valid; however, relative comparisons are permissible.



3. Soils: Substrate qualities and nutrient dynamics

Objectives. Soil conditions have a major influence on vegetation growth and on organisms that inhabit the rhizosphere of plants (e.g., amphipods, nematodes, microbes). Four variables are especially helpful in predicting the ability of a site to support a functional salt marsh: soil salinity, nitrogen dynamics, organic matter concentration, and redox potential. Soil salinities control seed germination and seedling establishment in the coastal wetlands (Zedler and Beare 1986). Concentrations that are either too high or too low will alter vegetation composition by restricting growth of even the most tolerant halophytes, in the case of extreme hypersalinity, or by allowing the invasion of cattails, bulrushes, and other glycophytes, in the case of prolonged periods of hyposalinity. Nutrient dynamics, organic matter, and redox conditions all interact to control plant growth rates, which in turn affect the consumers that live among the plant roots. Soils with low organic matter will have low nitrogen-fixation rates and low supplies of the main nutrient that limits plant growth. Soils with high organic matter will develop very negative redox potential, which may restrict the growth of some marsh plants; Cantilli (1989) showed that low redox affected the growth of pickleweed, but probably not that of cordgrass.

The patterns of salinity, nitrogen dynamics, organic matter accumulation, and redox potential vary in space and time. Wetland hydrology determines the chemical and physical nature of salt marsh substrate to a great extent (Mitsch and Gosselink 1986). For instance, the aerobic-anaerobic conditions resulting from regular tidal flooding of these soils have a profound effect on the biogeochemical cycles of nutrients, and consequently, on the marsh productivity. To date, there are only a few sites for which we have obtained detailed data on soil conditions. The theses of Swift (1988), Cantilli (1989), and Zalejko (1989), and the studies of Langis et al. (in press) provide a basis for understanding belowground dynamics of the coastal salt marshes.

Sampling strategy. All soil sampling is done randomly along predetermined 10-12 m transects. The transects should be along selected elevations, so that replicate samples are under a similar tidal regime. Samples subject to seasonal variation (soil or pore water nitrogen concentration, nitrogen-fixation, denitrification, soil or pore water salinity, redox potential, water content) should be taken quarterly.

High variances among replicate samples are more often the rule than the exception for chemical parameters of soils. This variability can be reduced by compositing several samples before analysis (Binkley and Vitousek 1989). In general, one should increase sampling as much as is practiable to improve the estimate of the mean. Soil analysis time is the usual limiting factor. Lloyd and McKee (1983) have suggested a statistical procedure for determining the optimal number of subsamples required by the level of confidence desired. The number of subsamples will depend on the degree of variability of the measurements for the sediments of the particular area. For example, Langis et al. (in press) obtained acceptable levels of variation with 4-6 soil cores randomly selected along pre-established transects of 10-12 m.

Bulk density. This parameter represents the weight of soil per unit volume including pore spaces. It is useful when the density differences of different soils must be accounted for. In which case, properties such as percent organic matter, total nitrogen or moisture can be overestimated in organic rich sediments when expressed on a dry weight basis because of the relative low weight of organic matter (Allison 1973).

Bulk density is obtained by pressing into the soil a thin-walled soil can with a cutting edge. The soil is then sliced smooth at the open end and oven-dried to constant weight at 105°C. Bulk density = mass of soil in tube/volume, expressed as g dry-wt/cm^3 (Richards 1954).

Water content is measured as weight loss upon oven drying divided by dry weight of the soil sample. The soil sample is dried to constant weight at 100-110°C (Gardner 1986). Organic soils have higher water-holding capacities than mineral soils (except for clays).

Particle size of marsh soils should be assessed in the traditional method used by terrestrial ecologists, identifying sand, silt, and clay percentages using a hydrometer (Gee and Bauder 1986), rather than the detailed size differentiation used by marine ecologists (Emery settling tube and phi values). Soils are characterized by type, with natural salt marshes likely falling in the clay to clay-loam types. At Tijuana Estuary, lower marsh soils were generally clayey, with 2-8% sand, 23-32% silt, and 46-58% clay (Zedler et al. 1980).

Swift (1988) compared soil texture in man-made and natural marshes at Sweetwater River Wetland Complex and found coarser texture in soils of the marsh that had been graded to expose lower-elevation strata. Paradise Creek (natural marsh) soils were clay loams, while those of Caltrans Connector Marsh (man-made) ranged from loam to sandy loam. These differences in texture, along with differences in soil organic matter (lower in the man-made marsh) appeared to be important to soil functioning, as there were also lower rates of nitrogen fixation and less evidence for sulfate reduction.

Soil salinities are assessed to help explain vegetation patterns and to track the influence of freshwater inflows. In all cases, measurements are made from replicate sampling stations. Soil salinity is measured monthly, near the water salinity sampling locations. Soil salinities are also measured at the permanent vegetation transects at the time of plant censusing, and in sites of special interest, such as planting locations for the salt marsh bird's beak and areas where weedy vegetation is invading.

Soil cores are easily removed with a 2.5-cm-diameter soil tube (available from Forestry Suppliers), which is inserted about 20 cm into the soil. A subsample of the core is then analyzed for soil salinity. A 5-cm segment from the 0-10-cm depth characterizes the upper root zone; and 20-30 cm the deeper root zone. If the marsh soil is saturated with water, the salinity measurement can be made immediately in the field, by expressing a drop of soil water onto a salinity refractometer (use one that is temperature-compensated, with a range of 0-150 ppt; available from most scientific supply houses). Soil water is expressed by loading a 10 cc plastic syringe (without needle) with 2 layers of #2 Whatman filter paper that has been cut into 12-mm-diameter circles (punch from larger sheets using a half-inch die). The wet soil is loaded by hand; the plunger is inserted, and a drop of water is forced onto the refractometer. Readings are in ppt.

If the soil is too dry, the subsample is stored in a plastic bag (Whirlpak™ bags are handy) and returned to the lab for artificial saturation with deionized water. The standard method for preparing saturated soil pastes should be followed (refer to Richards 1954) and salinity measured with a conductivity meter (e.g., Lab-Line mho meter). This method is time-consuming, but soil samples can be refrigerated and processed in batches. The conductivity results are not directly comparable with those from saturated soils, but salinities from all pastes are comparable with one another.

For comparison of the salinity of saturated soil pastes with other refractometer measurements taken in the field, water from the soil paste can be expressed through filter paper and measured with the refractometer. Results can be viewed as the soil salinity that would occur following rainfall.

Reference data on soil salinity are available for several wetlands in San Diego County. A major factor that influences soil salinity is closure of the ocean inlet, as illustrated by comparing data from Los Peñasquitos Lagoon (Figure 3.1), which is often a closed lagoon, with data from Sweetwater River Estuary (Figure 3.2) and Upper Newport Bay (Figure 3.3), which are always open to tidal flushing. Los Peñasquitos Lagoon soils become very hypersaline following evaporation and are measurably diluted following rainfall. The effects are stronger for surface soils (0-5 cm) than deeper in the profile (45-50 cm). In contrast, soils of tidally flushed wetlands are mildly hypersaline for most of the year. With a nontidal system, surface soil salinities are more variable than with the ameliorating influence of tidal waters.

Long-term data on soil salinity have been recorded at Tijuana Estuary, as part of the salt marsh monitoring program of PERL (Figure 3.4). Soil salinity has been measured in April and in September for 10 years, including changes following a major flood year (1980) and estuarine closure during a drought year (1984).

Measurements to be done on pore water. Since dissolved nutrients are readily available to plants, it is advantageous to sample pore water. These samples are representative of chemical conditions in the sediments. In anaerobic sediments, dissolved nitrogen is mostly under the NH_4^+ and dissolved organic forms, NO_3^- being a transient species. Ammonium and SRP (soluble reactive phosphorus) concentrations reflect exchange equilibria between dissolved and solid phases, since NH_4^+ and PO_4^{-2} are adsorbed on fine organic and clay particles (Avnimelech et al. 1983).

Pore water chemistry can be evaluated in a convenient way by the use of wells that can be permanently installed in the sediments. Water should be sampled as soon as possible after a high tide so that sediments are sufficiently saturated. To obtain a fresh sample, each well must be emptied with a syringe fitted with a vinyl tube (long enough to get to the bottom of the well) and allowed to refill before collecting the pore water sample.

Parameters such as pH, redox potential and salinity can be measured directly in the well and samples can be taken in the laboratory for chemical analyses. Ammonium concentrations can then be measured with the hypochlorite-nitroprusside method and NO_3^-+NO_2^- by the cadmium reduction method (APHA 1986). The depth of sampling is adjusted through the location of the slits and by inserting the wells at different depths. It is possible to collect water from a narrow or broad range of depths using such wells. Water samples for chemical comparisons must be collected at similar tidal levels since Agosta (1985) found that NH_4^+ concentrations tend to decrease as the water table goes down and to increase as the water table comes back up. This would especially apply to creek banks where the level of the water table is strongly affected by the tidal action.

The wells are constructed from plastic (PVC) pipes (30-cm long, 2-cm inside diameter, 2.5-cm outside diameter), in which a series of vertical slits is cut at 2-cm intervals, from the bottom to approximately 17 cm. The slits are covered with a piece of nylon screen (Nitex™, 100 to 200 µm) held in place with a sleeve made from a thin PVC tube of dimensions similar to the well. Since small particles will get through the screen, particles should be allowed to settle, or it might be necessary to centrifuge or filter the sample with Whatman™ GF/C filter paper before analysis.

Figure 3.1. Substrate salinity at Los Peñasquitos Lagoon, a system that is often closed to tidal flushing. Evaporation of sea water leads to hypersaline soils, while impounded rainfall and streamflow reduces salinity. Data are means (and ±1 s.e.) of samples at the surface (0-5 cm) and at depth (45-50 cm) for n=21 stations. Data are from Eilers (1981).

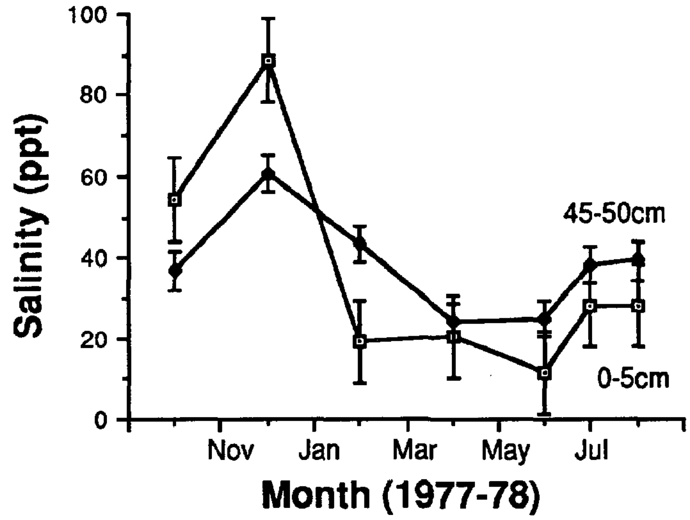


Figure 3.2. Substrate salinity at Sweetwater River Estuary, a fully tidal salt marsh adjacent to San Diego Bay. Data are means (±1 s.e.), for n=31 stations. Data are from Eilers (1981).

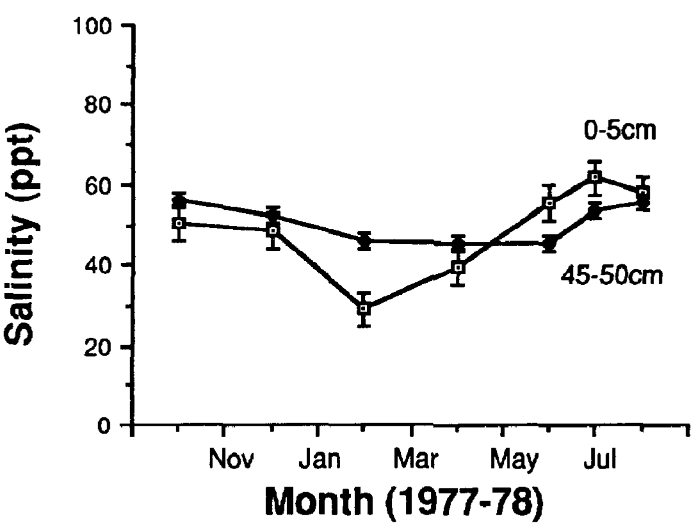


Figure 3.3. Substrate salinity at Upper Newport Beach, a fully tidal salt marsh. Data are means (±1 s.e.), for n=25 stations. Data are from Eilers (1981).

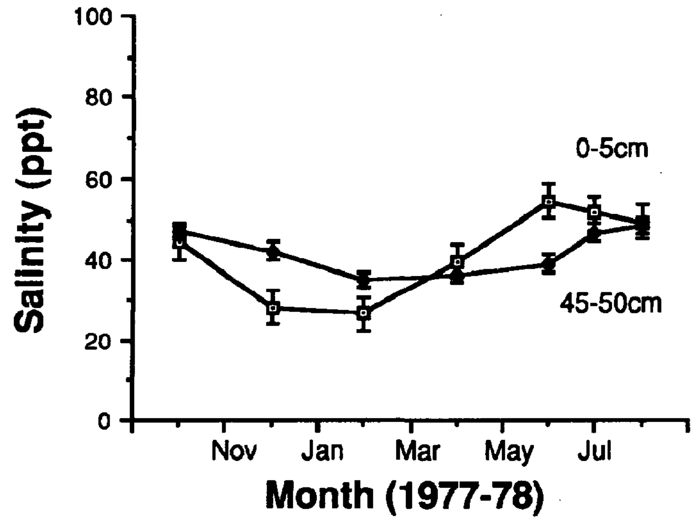
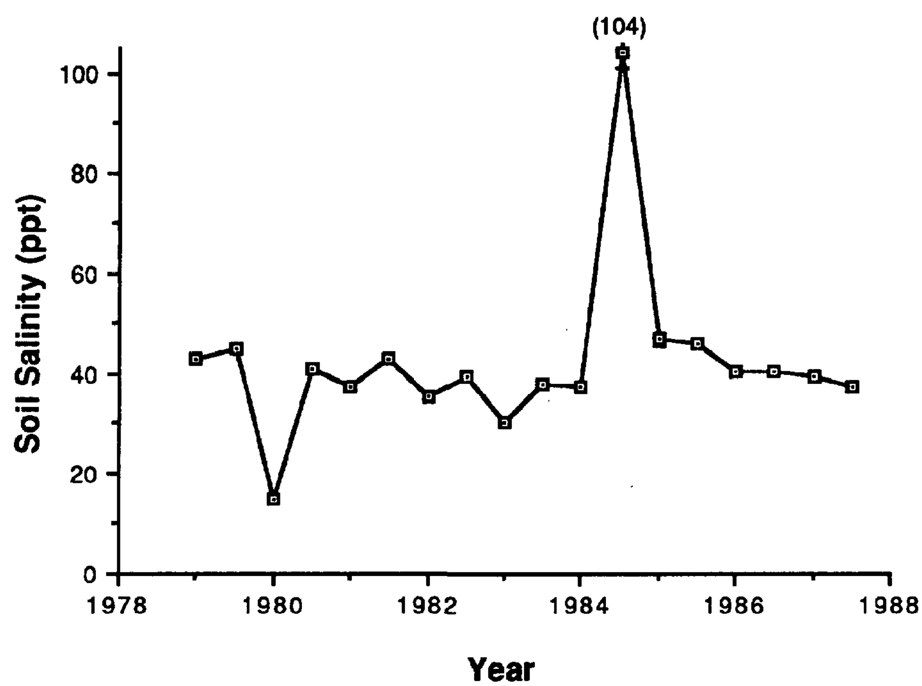


Figure 3.4. Interstitial soil salinity (0-10 cm) at Tijuana Estuary. Data are from the lower marsh, at 102 sampling stations within the 1979 distribution of cordgrass, for April and September sampling periods. Error bars (±1 s.e.) are generally too narrow to show.

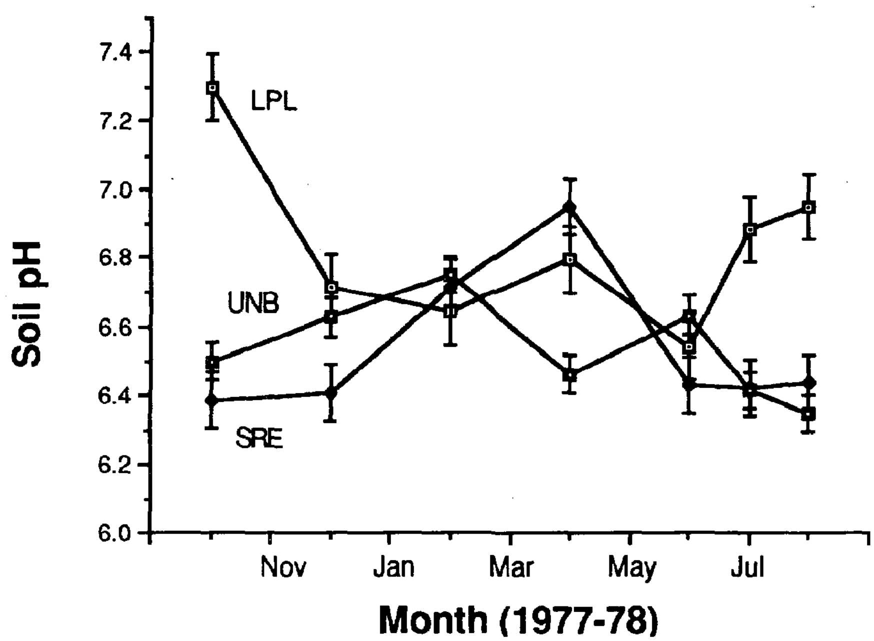


Soil pH can be measured directly in pore water wells, using a pH meter and a combination electrode (cf. Figure 3.5). If pore water wells are not used, the electrode can be inserted gently into a moist soil sample. If soil samples are too dry, they could be sealed in a plastic bag and returned to the laboratory where this measurement can be made on soil paste (see section on soil salinities). Soil pH could become a major concern in the case of acid sulfate soils. These soils can become extremely acidic following the oxidation of sulfides to sulphates and sulfuric acids. This situation occurs when tidal inundation is stopped and leaching of sulfates and sulfuric acid by rain is impeded by a high clay content (Linton 1969). Soil pH can vary by as much of 2 units within a tidal cycle, because of water infiltration or benthic biological activity (Wolaver et al. 1986). Values lower than pH 4 are detrimental to salt marsh plant establishment, since Broome (1987) noted no suvival of vegetation planted in soils of pH<3.

Redox potential. This measurement is important because of it plays an important role in the biogeochemical cycle of nitrogen and sulfur, and affects the mobility of heavy metals. As for pH, it can be easily measured by inserting a redox probe directly in the pore water sampling well.

Values obtained with the redox electrode must be standardized to measurements obtained with the standard hydrogen electrode by adding the appropriate correction factor. The correction factor is the difference between values measured with ZoBell's solution (0.003 M potassium ferricyanide, 0.003 M potassium ferrocyanide and 0.1 M potassium chloride) and its theoritical Eh of +430 mV at 25°C (ZoBell 1946).

Figure 3.5. Soil pH measured in bore holes at Los Peñasquitos Lagoon (LPL, n=21 stations), Sweetwater River Estuary (SRE, n=31 stations), and Upper Newport Beach (UNB, n=25 stations). Data are means (±1 s.e.). Data from Eilers (1981).



Extractable NH_4^+ and NO_3^- $+NO_2^-$. These inorganic forms of nitrogen give an estimate of potentially available nitrogen in the soil. To measure, put 10 g (wet wt) of fresh soil sample of in a 125 ml Erlenmeyer flask; add 100 ml 2 M KCl; set on a wrist-action shaker for 1 h; filter through Whatman™ no. 1 (the filtrate can be kept frozen until analysis); analyze for NH_4^+ with the automated phenate meth-od and for $NO_3^-+NO_2^-$ by the cadmium-reduction method (APHA 1986). Artificial seawater should be used to prepare blanks and standards. In the characteristically anaerobic sediments of salt marshes most of the extractable N will be under the form of NH_4^+, NO_3^- contributing very little to the N pool.

Total nitrogen. This is a measurement of the total nitrogen in the soil and includes the organic and inorganic forms. Nitrogen is either measured as NH_4^+ after Kjeldahl digestion (APHA 1986) or directly with a CHN analyzer.

Percent organic matter. Both combustion and chemical oxidation methods have been used to measure soil organic matter (O.M.) content. Weight loss upon combustion in a muffle furnace overestimates O.M. when clay minerals and/or carbonates are destroyed at high temperatures. Swift (1988) measured O.M. or wetland soils by combusting samples at 700°C for 1 h and 400°C for 3 h. Values obtained at 700°C were consistently higher (Table 3.1).

Two wet digestion techniques are recommended to avoid the problems of carbonate or clay degradation. The rapid oxidation technique (Sims and Haby 1971) has provided satisfactory results. Briefly, 10 ml of 1N $K_2Cr_2O_7$, followed by 20 ml of concentrated H_2SO_4 are added to a 1-g (dry wt) soil sample in a 125 ml Erlenmeyer flask. The slurry is swirled and allowed to react for 20 min, brought to a volume of 100 ml with distilled H_2O and centrifuged (at ca.500-1000G) for 15 min. Absorption of the supernatant is measured at 600 nm on a spectrophotometer. Absorption values are then compared to a series of standards (1.5 to 7% O.M.). Standards are prepared by adding calculated amounts of sucrose to ignited sediments (750°C, 4 h). The main drawback of the method is that some of the refractory organic substances will resist the digestion.

Organic carbon. The refractory organic carbon fraction could better be accounted for with a second technique, the modified Mebius procedure (see Yeomans and Bremner 1988), where a more thorough oxidation is obtained by digesting the sample at 170°C with potassium dichromate and values of % organic carbon are obtained directly by titration.

A CHN analyzer could also be used to get organic carbon measurements simultaneously with total nitrogen on samples with unmeasurable levels of inorganic carbon. If inorganic carbon is present at a significant level (see Nelson and Sommers 1982 for a discussion of methods) it is advisable to use the modified Mebius procedure (Yeomans and Bremner 1988). The Sims and Haby procedure could also be used provided that standards are expressed as % O.C. instead of as % O.M.

Nitrogen fixation could represent an important source of available nitrogen for the generally nitrogen-limited salt marsh vegetation. The most used and most straightforward method involves the use of the acetylene reduction reaction, where the enzyme dinitrogenase is capable of reducing C_2H_2 as well as dinitrogen (Hardy et al. 1968; Casselman et al. 1981). In this technique, C_2H_2 is added to incubation vessels and dinitrogenase activity is monitored. Although this method must be calibrated with an ^{15}N tracer to obtain absolute values, comparisons can be made on the basis of nmoles C_2H_2 reduced per unit soil.

Table 3.1. Soil organic matter in the natural wetland remnant, Paradise Creek Marsh, and the man-made Connector Marsh at San Diego Bay. Data are for soils seived with a 2-mm mesh screen to remove roots (from Swift 1988).

		400°C for 3 hr		700°C for 1 hr	
	Depth	0-5 cm	5-10 cm	0-5 cm	5-10 cm
Paradise Creek	mean	**4.90**	**4.86**	**8.14**	**9.24**
n=3	s.e.	(0.20)	(0.48)	(0.24)	(0.48)
Connector Marsh	mean	**2.21**	**1.61**	**4.34**	**4.17**
n=6	s.e.	(0.16)	(0.13)	(0.34)	(0.27)

For measuring nitrogen fixation, we recommend the following procedure adapted from Zalejko (1989): In the field, soil cores are taken for root zone N-fixation (10-cm deep by 8-cm diameter) and for surface N-fixation (1 cm deep by 3 cm in diameter for surface N-fixation). Cores are placed in a 1-L mason jar and tightly capped with a lid (equipped with a serum stopper). It is important to clean dirt off rims since jars must be air-tight.

The cores are incubated overnight in the dark at 22°C, to allow microbial populations to adapt. The next morning, acetylene and ethane (as an internal standard to account for leakage) are added to jars with a syringe, yielding an atmosphere of 10% and 0.01%, respectively. Concentrations of C_2H_4 and C_2H_6 are measured with a flame ionization detector on a gas chromatograph. The headspace gases are mixed by pumping the syringe plunger several times. Gas samples are withdrawn from the jar head space on several occasions (after 1 to 5 h of incubation). Samples (1 ml) are injected (injector temperature: 85°C) onto a 2 m Porapak™ N column. Pure nitrogen is used as the carrier gas at a flow-rate of 30 $ml \cdot min^{-1}$, the oven temperature is 50°C and the detector temperature 200°C. Measurements are done on peak areas.

Nitrogen mineralization. Estimates of nitrogen mineralization rates in the field can be obtained with the buried polyethylene bag technique (Eno 1960). Because plant uptake and leaching of nitrogen is prevented, net mineralization rates can be estimated as the increase in NH_4^+ and NO_3^-+NO_2^-(Pastor et al. 1984). Sediment cores (8-cm depth x 5-cm diameter) are placed into 0.03 mm thick polyethylene bags; the bags are tied and replaced in their respective holes and covered with approximately 2 cm of sediment. At the same time, another soil core, adjacent to the plastic-enclosed core, is collected and immediately taken to the laboratory for analysis of extractable NH_4^+ and NO_3^-+NO_2^-. These polyethylene bags are impermeable to water and permeable to gases such as CO_2 so that constant moisture content can be assured during the incubation while permitting gas exchange (Gordon et al. 1987). After 14 days, the bags are collected and taken back to the laboratory for analysis. Increases in concentrations of extractable NH_4^+ and NO_3^-+NO_2^- (an estimate of net nitrification) are then calculated to estimate rates of mineralization.

4. Vegetation composition and growth

Objectives. Wetland vegetation is the most obvious and straightforward indicator of habitat condition. Surveys of vegetation are needed to document the success of plant growth as well as to assess the site's potential for supporting animal populations.

The species composition of habitats within an area allows one to read the history of the site. Changes in vascular plant distributions lag behind environmental changes, because most species are limited in their ability to become established even when the habitat is appropriate. Thus, the vegetation is an integrator of long-term conditions, more than a measure of current events. The presence of cordgrass indicates a long history of good tidal flushing, but the species may persist for many months or even years after such conditions have changed. The presence of cattails in a salt marsh indicates that excess freshwater inflows occurred at some time in the past, but does not prove that flows are currently being augmented. If a site has few species present, there are two possible interpretations--the habitat may be poorly suited for salt marsh development, or the site may provide suitable habitat but not yet support a diverse vegetation due to isolation from propagules or insufficient time for establishment.

The plant species composition dictates the suitability of sites for a wide variety of animals. Insects are among the most host-specific species, but several wetland birds are also restricted to areas dominated by specific plants. Additional linkages between vascular plants and animals, especially those in the soils and and root zone (rhizosphere), still await discovery.

Vegetation maps. Aerial photographs are obtained to determine overall vegetation coverage in each marsh. General attributes to obtain are the total area of each constructed marsh, the amount of open (unvegetated space), and the area of each type of vegetation (e.g., lower intertidal marsh dominated by cordgrass, mid-intertidal marsh dominated by mixed succulents, high-intertidal marsh dominated by various species and with potential habitat for salt marsh bird's beak). Next, representative subunits of each vegetation type in both the constructed and reference wetlands are selected for detailed sampling to determine if the plant communities are similar.

Approximate elevations are determined from a topographic map (one-foot contour map needed) and used to locate and set the length of each transect (50-m transects are useful, but shorter lengths may be necessary in variable topography). To characterize the high marsh-upland transition area, the transect should parallel the elevation band that contains that habitat. About 40-50 stations should be established in each area to be compared. Each transect should be indicated on an aerial photo overlay. Elevations should then be surveyed at each sampling station. Each transect should have a number, and the number of sampling stations on each transect should be recorded on the overlay.

Sampling stations should be located at regular (5-m) intervals along permanent transects. Sturdy wooden stakes should be numbered with marine paint and placed so they barely protrude above the canopy. Tall stakes attract raptors that prey on water-associated birds (especially young chicks), and it would not be desirable to place predator roosts in a marsh used by clapper rails.

Quantitative sampling for vascular plant species composition. A long-term sampling program at Tijuana Estuary (1979-present) provides the most extensive data set for salt marsh species composition. Comparable sampling methods are thus recommended. The sampling program records species presence (for frequency of occurrence data), visual cover estimates for all species, and more intensive analysis of cordgrass, which is often a restoration target species. The most important feature for comparing occurrence data is quadrat size; data from different size quadrats are not comparable, as larger quadrats encounter more species. Quarter-square meter quadrats are suitable for salt marsh vegetation, because several individuals can be found within that area. Quadrat shape should also be held constant. Circular quadrats were chosen for Tijuana Estuary, because an initial objective was to understand interspecific interactions. The species occurring together in a 0.25-m^2 circular quadrat have intermingling roots. Thus, data on species that occur together are useful in evaluating interactions. Additional useful measurements are canopy height and data on flowering and/or fruiting by species.

To determine species composition, cover, and canopy heights, permanent sampling locations (quadrats along transects) are established and marked for elevation. These are sampled in late August or September for presence and cover of each species and for heights of cordgrass stems, within 0.25 m^2 circular quadrats.

Additional data are useful for cordgrass, which has a growth form that allows easy measurement of culm height. In dense stands, 0.10 m^2 quadrats are adequate to give reliable estimates of the mean cordgrass growth. Heights are summarized to provide total stem length (sum of all heights, which is a good estimator of aboveground biomass), average height, and density (number of stems).

Other species are less amenable to height measurement or density counts, because of their trailing and branched habit and the difficulty of deciding what constitutes an individual. For this reason, cover data are preferred. Because the sum of cover of individual species may exceed 100%, a separate visual estimate must be made for total vegetative cover.

Six cover classes have been used in the Tijuana Estuary monitoring program. Frequency histograms of cover classes are readily compared with the Kolmogorov-Smirnov two-sample test. Mean cover can be estimated by using the midpoints of each cover class. Because of the imprecision involved in estimating cover, differences of less than 25% between two sites are not meaningful.

Cover class	Midpoint of cover class
>0-1%	0.5%
1-5	3.0
6-25	18.5
26-50	38.5
51-75	63.5
76-100	88.5

Cover of vegetation is compared more precisely by sampling canopy intercept along the transect lines. The intercepts of each species and of bare space are recorded within 4- or 5-m segments of each transect line. The smallest unit of intercept recorded is 1 dm. Lines are placed along elevation contours, and the meters of cover within 4-m segments of the line are recorded to the nearest decimeter. A simple comparison of man-made and natural wetlands uses a cumulative frequency histogram of the number of segments with 0, 0.1-1.0, 1.1-2.0, 2.1-3.0, and 3.1-4.0 meters of cordgrass cover (Figure 4.1, data from Swift 1988). In the cumulative histogram, the tally for the first cover class is recorded, then the tally for the second class is added to the first, etc., until all tallies are included in the last point of the graph. From this summary of the data, it is clear that the man-made marshes have far more area of zero or low cordgrass cover than the reference wetland, Paradise Creek. The man-made site that most closely approximates cordgrass cover in the natural marsh is Nursery 1, which was in its third growing season at the time of Swift's survey.

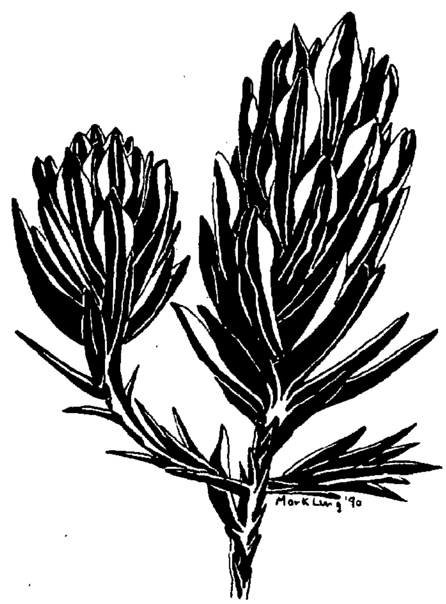
Censuses of target species (desirable and undesirable) should be made during the period of time when they are most conspicuous. High marsh is examined in April to locate and census salt marsh bird's beak patches (*Cordylanthus maritimus* ssp. *maritimus*) and associated (potential) host species. The size of each patch is measured (maximum diameter and the diameter perpendicular to it), and counts of individuals are made. Soil salinities at 10- and 30-cm depth are measured at each patch. If rainfall has been late, April may be too soon to census patches, and searches may need to be repeated later.

Patches of weedy species are located and sampled in April, as many of the exotics are short-lived annuals. The area of weedy vegetation is measured as above, and soil salinities recorded for 10- and 30-cm depths.

An additional plant population to be monitored more closely is the rare annual goldfields, *Lasthenia glabrata*, which may be found at the edges of seasonal pools or salt pannes that impound rainfall. The population at Los Peñasquitos Lagoon has been monitored for several years (Nordby, SDSU, unpub. data), by locating and measuring the size of patches, plus obtaining density estimates using the nearest-neighbor method and by direct counts of densities using 0.25 m^2 quadrats and 0.10 m^2 quadrats.

Reference data: Tidal marsh community composition: Ferren (1985) gives the most complete list of salt marsh plants, based on Carpinteria Salt Marsh. His cumulative list of salt marsh, brackish marsh, and transition to upland includes 38 species (Table 4.1).

The regional distribution of a subset of these species is given in Table 4.2. This regional comparison shows that species richness relates to the degree of tidal flushing--wetlands with a long history of tidal flushing have most of these species.



Salt marsh bird's beak
Cordylanthus maritimus
ssp. *maritimus*

Figure 4.1 The cumulative percent of 4-m intervals with 0-to-4 meters of cordgrass cover near Sweetwater Marsh, San Diego Bay in summer 1987. Each transect followed the elevation of maximum cordgrass establishment. The Connector Marsh sites were graded and opened to tidal flow in fall 1984 and planted with cordgrass in Jan.-Mar. 1985. Nursery 1 was graded and planted in July-Nov.1983. Paradise Creek is a natural marsh remnant just upstream of the Connector Marsh.

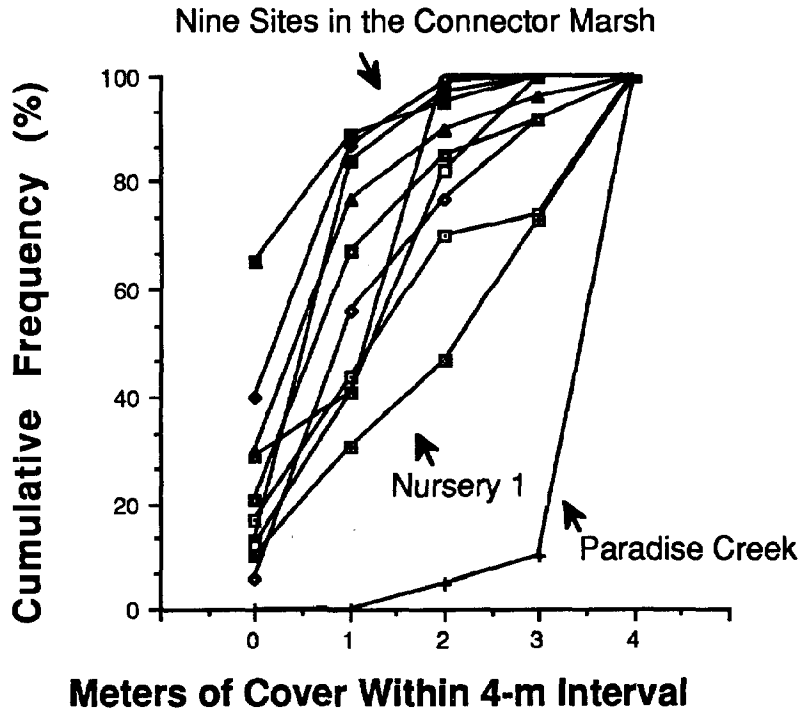


Table 4.1. Native plant species list for "estuarine wetlands" of the Carpinteria salt marsh (from Ferren 1985). *=not seen by Ferren; possibly extirpated from this wetland.

Anemopsis californica
Arthrocnemum (=Salicornia) subterminale
Aster subulatus var. *ligulatus*
Atriplex californica
A. lentiformis ssp. *breweri*
A. patula ssp. *hastata*
A. watsonii
Baccharis douglasii
B. pilularis ssp. *consanguinea*
**Carex praegracilis*
**Chenopodium macrospermum* ssp. *farinosum*
C. strictum
Cordylanthus maritimus ssp. *maritimus*
Cressa truxillensis ssp. *vallicola*
Cuscuta salina
Distichlis spicata ssp. *spicata*
Euthamia occidentalis
Frankenia grandifolia ssp. *grandifolia*
Heliotropium curassavicum ssp. *occulatum*
Hordeum depressum
Hymenolobus procumbens
Isocoma veneta var. *vernonioides*
Jaumea carnosa
**Juncus acutus* var. *sphaerocarpus*
J. bufonius
Lasthenia glabrata ssp. *coulteri*
Limonium californicum
Monanthochloe littoralis
Salicornia virginica
Scirpus californicus
S. maritimus
**S. pungens*
Spergularia macrotheca var. *macrotheca*
S. marina
Suaeda calceoliformis
S. californica var. *pubescens*
Triglochin concinna
Typha domingensis

Table 4.2. Presence of selected native plants in southern California coastal salt marshes. X=extant, based on literature or personal observations; *=reintroduced, long-term status uncertain; #=recently declined from greater abundance. Bold type indicates fully tidal systems (ocean inlets are rarely closed). [Please forward any new information for these species lists to PERL.]

Species codes: *Fp=Frankenia palmeri; Cm=Cordylanthus maritimus* ssp. *maritimus; Bm=Batis maritima; Sb=Salicornia bigelovii; Sf=Spartina foliosa; Lg=Lasthenia glabrata; Tc=Triglochin concinnum; Aw=Atriplex watsonii; Lc=Limonium californicum, Ct=Cressa truxillensis; Ml=Monanthochloe littoralis; Se=Suaeda esteroa; Cs=Cuscuta salina; Ja=Juncus acutus; Sv=Salicornia virginica; Fg=Frankenia grandifolia; Ds=Distichlis spicata.*

Coastal Salt Marsh	Fp	Cm	Bm	Sb	Sf	Lg	Tc	Aw	Lc	Ct	Ml	Se	Cs	Ja	Ss	Jc	Sv	Fg	Ds	#
Sweetwater Marsh	X	X	X	X	X		X	X	X	X	X	X	X	X	X	X	X	X	X	18
Tijuana Estuary		X	X	#	X	X	X	X	X	X	X	#	X	X	X	X	X	X	X	18
Mugu Lagoon		X	X	X	X	X	X	X	X		X	X	X	X	X	X	X	X	X	17
Anaheim Bay		*	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	16
Upper Newport Bay		X	X	X	X		X	X	X		X	X	X	X	X	X	X	X	X	16
Santa Margarita Estuary				X		X		X	X	X	X	X	X	X	X	X	X	X	X	14
Bolsa Chica Wetland			X	X	X			X	X	X	X	X		X	X	X	X	X	X	14
Mission Bay Reserve			X	X	X		X		X		X	X	X		X	X	X	X	X	13
Carpinteria		X				X	X	X	X	X	X		X		X	X	X	X	X	13
Goleta Slough						X	X	#	X	X	X	#	X		X	X	X	X	X	13
Los Peñasquitos Lagoon						X		X	X	X	X	X	X	X	X	X	X	X	X	13
San Dieguito Lagoon					*	X			X	X	X	X		X	X	X	X	X	X	12
Batiquitos Lagoon						X			X	X	X			X	X	X	X	X	X	10
Agua Hedionda Lagoon										X	X	X	X	X	X	X	X	X	X	10
Ballona Wetland										X					X	X	X	X	X	6
San Elijo Lagoon										X					X	X	X	X	X	6
Deveraux Lagoon										X			X			X	X	X	X	6
Santa Clara R. Estuary										X				X		X	X	X	X	6
San Luis R. Mouth														X		X	X	X	X	5
Las Flores Marsh															X	X	X	X	X	5
McGrath Lake														X		X	X	X	X	5
Malibu Creek													X			X	X	X	X	5
San Mateo Marsh													X				X	X	X	4

For a few salt marshes in southern California and northern Baja California, descriptions are available in the published literature. See Ferren (1985) for data on Carpinteria Marsh, Onuf (1987) for Mugu Lagoon, Schreiber (1981) for Ballona Wetland, Vogl (1966) for Upper Newport Bay; Zedler and Beare (1986) for San Diego River Marsh, Zedler (1977) for Tijuana Estuary, and Neuenschwander et al. (1979) for San Quintin Bay, Baja California. Broader discussions of the California salt marshes appear in Macdonald and Barbour (1974) and Macdonald (1977, 1988). Data for additional wetlands appear in less widely-circulated reports. The US Fish and Wildlife Service has provided preliminary results for Santa Margarita Estuary (Hollis et al. 1988), and the Topanga-Las Virgenes Resource Conservation District has released a baseline survey for Malibu Lagoon (Manion and Dillingham 1989). Additional wetlands under study by PERL/SDSU researchers include Ballona Wetland, Los Peñasquitos Lagoon, and Sweetwater River Wetland Complex. General descriptions of many of the region's coastal wetlands have been published by the California Dept. of Fish and Game, as part of a Wetland Resources Inventory, carried out in the 1960's and 70's.

The detailed studies of marsh vegetation are useful in showing how different species respond to reductions in tidal flushing. The composition of Santa Margarita Estuary (Table 4.3), which is often closed to tidal flow in summer, is in strong contrast with the species-rich marsh at Tijuana Estuary (Table 4.4). There are fewer species overall, and *Salicornia virginica* has very high dominance. Tijuana Estuary, in turn, lost some of its richness during the 1984 nontidal period of 8 months. At Santa Margarita Estuary, three pickleweed habitat types were distin-guished by Hollis et al. (1988), but the tabular data show considerable overlap in species composition among sites that appeared different to the eye. The results are useful in showing transect-to-transect variability in species abundance.

Species composition within salt marshes is related to elevation, which in turn indicates differences in inundation, salinity, and a host of other environmental factors. The following data were obtained in the September 1988 annual census of the Tijuana Estuary salt marsh, including 207 quadrats between 5-18 dm MSL (approximately 1.5-6 ft MSL, or 4.5-9 ft NGVD). Since most of the quadrats occur on the marsh plain, these data characterize the lower- and middle-marsh habitats best. The % frequencies are provided by elevation for all species that occurred in more than 10% of the quadrats. Occurrences of the less common species are listed. Cover (mean %, based on cover classes) is for quadrats of occurrence only (excluding 0's), so quadrat n is not constant. Elevation class 5-6 ranges from 5.0 to 6.9 dm, etc.

The summary data for occurrence and cover indicate different attributes of these species. Pickleweed (*Salicornia virginica*) is the most abundant, and it generally has high cover. Saltwort (*Batis maritima*) is widespread at lower elevations, but never has high cover. Shoregrass (*Monanthochloe littoralis*) is restricted to the high marsh, but this mat-forming grass generally has high cover. From the 1974 distributions at Tijuana Estuary, we have identified the elevation where percent occurrences are greatest and indicated the average of the maximum heights these species achieved within all quadrats of occurrence (Table 4.5).

Upper marsh and transition habitats have not been studied extensively. There are few areas of undisturbed transition-to-upland habitat . At Tijuana Estuary, one gradual slope adjacent to the salt marsh was sampled for species composition and results combined with the 1974 census data. The data indicate in general how far upslope many of the marsh species go (Table 4.6).

Table 4.3. Salt marsh vegetation at Santa Margarita Estuary from Hollis et al. (1988). Data in each column are meters of cover from one 100-m-long transect. Vegetation types are as named by Hollis et al. (1988). Species with less than 1% cover are indicated with an x.

	Salicornia/ Frankenia		*Salicornia/ Distichlis*			*Salicornia/ Upland*			Maritime Scrub	
Bare ground	1	0	21	30	26	9	0	6	7	22
Mean Height (cm)	47	41	28	23	38	24	34	31	55	124
Salicornia virginica	80	54	58	28	60	5	2	27		11
Jaumea carnosa								4		4
Suaeda esteroa								3		
Frankenia grandifolia	18	23			x		x	16		
Distichlis spicata		21	21	42	14			35		x
Atriplex watsonii		x								
Salicornia subterminalis						54	24			
Cressa truxillensis						x	2	4		
Atriplex patula								3		
Croton californicus									3	
Isocoma venetus							x		16	36
Opuntia sp.									7	
Heliotropium curassavicum								x		
Baccharus glutinosa										13
Ambrosia chamissonis								x		
Weedy exotics		x				32	69	4	66	13

Table 4.4. Species Composition of the Tijuana Estuary salt marsh, 1988.

Elevation class (dm)	5-6	7-8	9-10	11-12	13-18	Mean overall	Cover
Quadrat n	43	101	36	14	13	207	Varies
Species and Frequency	%	%	%	%	%	%	%
Spartina foliosa	74	39	22	0	0	**38**	**49**
Batis maritima	33	63	42	0	0	**45**	**20**
Jaumea carnosa	12	32	8	0	0	**19**	**16**
Salicornia virginica	86	96	86	39	0	**82**	**52**
Frankenia grandifolia	9	34	44	43	54	**32**	**22**
Monanthochloe littoralis	0	6	44	93	46	**20**	**47**
Salicornia subterminalis	0	0	14	64	85	**12**	**44**
Triglochin concinnum						**7**	**10**
Distichlis spicata						**7**	**17**
Limonium californicum						**2**	**6**
Cressa truxillensis						**5**	**10**

Additional species present in the marsh but not encountered in the 1988 sample
Salicornia bigelovii
Suaeda esteroa
Atriplex watsonii
Cordylanthus maritimus ssp. *maritimus*

Table 4.5. Summary data for peak frequency of occurrence and the average of maximum heights observed at Tijuana Estuary.

Species	Elevation of peak frequency (dm MSL)	Ave. max. height in cm (s.e.)
Spartina foliosa	3	60 (10)
Salicornia bigelovii	6	38 (10)
Batis maritima	6	24 (5)
Salicornia virginica	5	36 (9)
Jaumea carnosa	7	21 (4)
Suaeda esteroa	9	29 (8)
Frankenia grandifolia	10	23 (6)
Monanthochloe littoralis	11	24 (6)
Distichlis spicata	9	25 (6)
Limonium californicum	10	no data
Triglochin maritima	7	no data
Salicornia subterminalis	11	31 (6)
Cuscuta salina	9	no data
Cressa truxillensis	10	no data

Table 4.6. Elevation ranges of salt marsh species. Occurrences are for 2-dm elevation classes, combining the 1974 survey and the analysis of Site 1. Class 3 indicates 3.0-4.9 dm NGVD or 1.0-1.6 ft NGVD. Metric units are useful because the vegetation responds to elevation differences as small as 1 dm (10 cm).

Elevation class (relative to NGVD)															
decimeters	3	5	7	9	11	13	15	17	19	21	23	25	27	29	31
feet	1.0		2.3		3.6		4.9		6.2		7.5		8.9		10.0
Spartina foliosa	x	x													
Batis maritima	x	x	x												
Salicornia bigelovii	x	x	x	x											
Jaumea carnosa	x	x	x	x											
Suaeda esteroa	x	x	x	x											
Salicornia virginica	x	x	x	x	x	x	x	x	x	x					
Frankenia grandifolia	x	x	x	x	x	x	x	x	x	x	x	x			
Monanthochloe littoralis	x	x	x	x	x	x	x	x	x	x					
Distichlis spicata		x	x	x	x	x	x	x	x	x	x	x	x	x	x
Triglochin maritima		x	x												
Cuscuta salina		x	x	x											
Limonium californicum		x	x	x	x										
Salicornia subterminalis		x	x	x	x	x	x	x							
Atriplex watsonii			x	x	x	x	x	x	x						
Cressa truxillensis			x	x	x	x	x	x	x	x	x	x	x		

No salt marsh is stable; composition changes annually. From the long-term monitoring program, we know that cordgrass responds to varying environmental conditions (Zedler 1983, Zedler and Nordby 1986); it declined with hypersaline drought (1984) and recovered slowly thereafter. Annual pickleweed (*Salicornia bigelovii*), which was extremely abundant at the same elevations as saltwort in 1974 (Zedler 1977), was nearly extirpated during the 1984 drought; it may persist in one small patch. Sea blite (*Suaeda esteroa*) also declined in 1984, but persists more widely, although only as scattered individuals. In the 1984-1988 record, three species have been consistently common (*S. virginica, B. maritima, F. grandifolia*), 2 have not yet recovered from post-1984 declines (*J. carnosa, T. concinnum*), and 4 species have been consistently uncommon (*M. littoralis, S. subterminalis, D. spicata, L. californicum*). These latter four species are not rare at Tijuana Estuary; their low frequencies of occurrence simply reflect the low numbers of quadrats sampled at their "preferred" elevations. The strongest differences are seen by comparing the 1974 samples with those of 1988 (Table 4.7).

Even if composition changes little, the height and vigor of plants can be highly variable from year to year. The dynamics of *Spartina foliosa* and *Salicornia virginica* have been studied in detail and results published elsewhere (Zedler 1983, Zedler et al. 1986). Such information indicates that single samples of the vegetation do not fully characterize its condition.

Biomass and net aerial primary productivity (NAPP) of vascular plants. Harvesting vegetation to estimate standing crops or primary productivity is <u>not recommended</u> for southern California coastal marshes. The harvest method is too destructive for our remnant wetlands. Furthermore, the standing crops grossly underestimate production of the region's dominant species (Onuf 1987).

By comparing data from tagged branches with harvest data, Onuf determined that large amounts of leaf and stem material are lost from plants between harvests. Thus, estimates of NAPP from repeated harvests were too low by a factor of 2.3 for *Salicornia virginica*, 3.7 for *Jaumea carnosa*, 1.89 for *Limonium californicum*, and 2.9 for *Batis maritima* (Onuf 1987, p. 71). Additional errors are no doubt present due to the high heterogeneity of aboveground biomass that is present in the region's species-rich marshes. Unlike monotypic *Spartina alterniflora* marshes of the Atlantic and Gulf of Mexico Coasts, a quarter-square meter of southern California marsh vegetation may include 8 or 10 different species. Thus, obtaining representative samples of the biomass of individual species is nearly impossible. Combining biomass of all species for a total NAPP estimate is not recommended, because individual species reach peak biomass at different times; thus such harvest data would further underestimate productivity. In common with harvest studies in all regions and habitat types, net productivity estimates based on standing crops do not take into account losses due to herbivory between harvests.

Earlier studies (Winfield 1980, Zedler et al. 1980, Eilers 1981) erred in assuming that southern California harvest data could provide accurate estimates of salt marsh vascular plant productivity. Their data are best used as descriptions of the aboveground standing crop. Eilers' harvest data for 1977-78 have been reanalyzed for that purpose; sampling stations for the entire marsh have been pooled, and data for all species combined. The results for live and dead biomass (Figures 4.2-4.3) show that biomass accumulates in the absence of tidal flushing (i.e., in Los Peñasquitos Lagoon).

Table 4.7. Frequency of occurrence (% of quarter-square-meter quadrats sampled) at Tijuana Estuary in 1974 and 1988, comparing different sampling locations, but using the same dm elevation classes (6 = 6.0-6.9 dm NGVD). The larger % frequencies are in bold.

Elevation class (dm)	Year	6	7	8	9	10
n	1974	113	95	35	30	35
n	1988	37	77	24	25	11
Occurrence (%)						
Spartina foliosa	1974	24	0	0	0	0
	1988	**78**	**48**	**8**	**32**	0
Batis maritima	1974	**86**	**71**	31	0	0
	1988	30	61	**71**	**44**	**36**
Salicornia bigelovii	**1974**	**90**	**83**	**46**	**17**	**14**
	1988	0	0	0	0	0
Jaumea carnosa	**1974**	**42**	**88**	**60**	**53**	**51**
	1988	14	31	33	12	0
Suaeda esteroa	**1974**	**26**	**43**	**63**	**63**	**34**
	1988	0	0	0	0	0
Salicornia virginica	1974	69	66	80	70	46
	1988	**86**	**96**	**96**	**100**	**54**
Triglochin concinnum	1974	**6**	12	3	**10**	0
	1988	5	12	**8**	0	0
Frankenia grandifolia	**1974**	**23**	**54**	**77**	**87**	**91**
	1988	11	30	46	40	54
Limonium californicum	**1974**	**7**	**12**	**11**	**10**	**20**
	1988	0	1	0	8	0
Distichlis spicata	**1974**	**2**	**9**	**43**	**23**	0
	1988	0	4	4	4	9
Salicornia subterminalis	**1974**	**6**	**6**	**29**	**30**	**69**
	1988	0	0	0	0	46
Monanthochloe littoralis	**1974**	**19**	**32**	**63**	**90**	**97**
	1988	0	3	17	36	64
Cressa truxillensis	1974	0	0	0	0	0
	1988	0	0	0	**8**	**27**

Figure 4.2. Summary of live material harvested by Eilers (1981). Data are means (±1 s.e.) obtained from 0.10 m^2 rectangular quadrats. LPL= Los Peñasquitos Lagoon, n=21 stations; SRE= Sweetwater River Estuary, n=31 stations; UNB= Upper Newport Beach, n=25 stations.

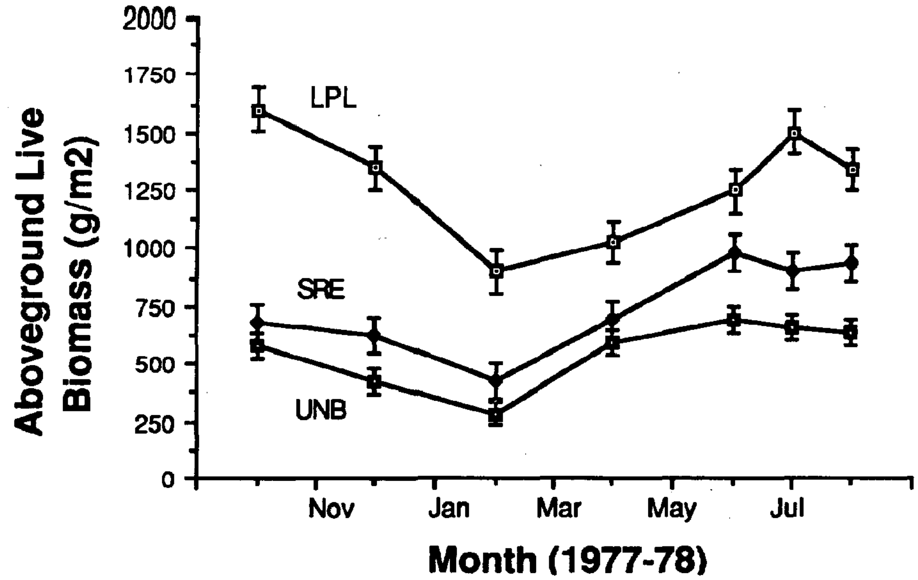
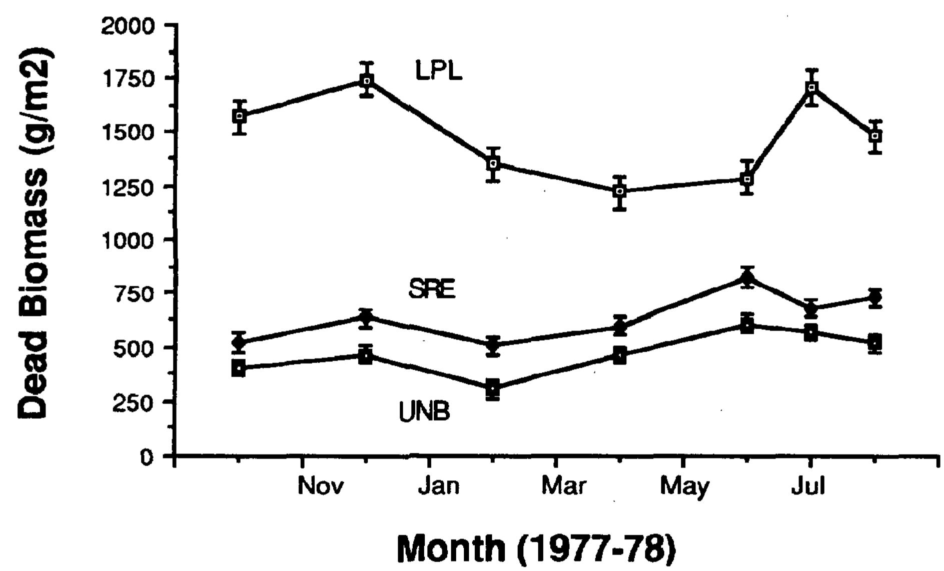


Figure 4.3. Summary of dead material harvested by Eilers (1981). Data are explained in Figure 4.2.



The large difference in the amount of dead biomass at nontidal LPL suggests that tidal flushing leads to faster decomposition and/or export, both of which are consistent with the findings of Winfield (1980). Additional historic data on standing crops appear in Winfield (1980) for Tijuana Estuary and in Zedler et al. (1980) for Tijuana Estuary, Los Peñasquitos Lagoon, and San Diego River Marsh.

There is a method for measuring vascular plant productivity using individual leaves or branches and assessing carbon dioxide uptake over short intervals of time (minutes). Portable systems are available (ca. $15,000) for measuring photosynthesis using infrared gas analysis to indicate changing concentrations of carbon dioxide in chambers that enclose intact leaves. However, photosynthetic rates for small portions of the plant must be combined with estimates of total leaf area to obtain net productivity rates for an area of marsh. Thus, some measure of biomass is still needed, along with frequent measurements to account for seasonal changes in rates and regressions to determine how rates vary with daily inundation and temperature. Such measures are best used to compare functional differences of different plants or habitats at single points in time.

In our opinion, it is not practicable to measure vascular plant productivity. While various methods can be employed, they are too destructive and do not yield data of sufficient accuracy to justify the damages to the site. Data on cover and height are more appropriate.

Algal productivity. Measurements of algal productivity are likewise problematical. Yet this component is probably very important to the food base, not only because of high growth rates, but also because of high digestibility (Zedler 1980, 1982a). Several kinds of algae contribute to wetland productivity. Phytoplankton and macroalgal mats become abundant in channels and tidal creeks, especially where tidal flushing is minimal (Rudnicki 1986, Fong 1986). Thick mats of filamentous blue-green algae and diatoms grow on the moist intertidal soils of the salt marsh and are especially productive in summer (Zedler 1980, 1982b).

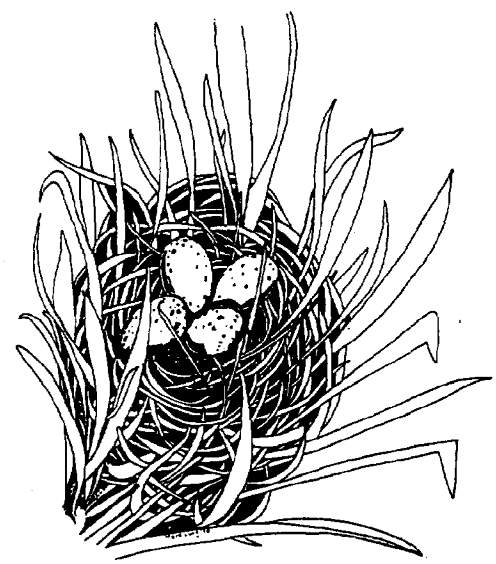
Estimating algal productivity rates is most easily accomplished by measuring the amount of oxygen evolved by fronds or mats placed in aquatic chambers (e.g., as carried out in the field by Zedler 1980). The oxygen concentration is measured initially and after about an hour's incubation time. Comparison of changes in light and dark chambers yields estimates of gross primary productivity. These short-term rates can then be used to estimate longer-term contributions of algae by relating gross photosynthetic rates to light regimes. The methods provide valuable comparisons of different algal types and different habitats, but several errors develop when calculations of annual productivity are attempted (Zedler 1980). In addition, sampling is very destructive, especially if one attempts to characterize a large number of habitats. Sampling must be frequent, as algal biomass and photosynthetic rates differ from week to week (with spring- and neap-tide inundation regimes), and a large number of replicates (for both light and dark chambers) is needed to obtain reliable results.

Results for Tijuana Estuary (Zedler 1980) for a year of good tidal flushing (1977) indicated that algal mats beneath the marsh canopy were highly productive, compared to Atlantic Coast wetlands, where taller, denser vascular plant canopies are present. While good data relating productivity rates to algal biomass or chlorophyll content are lacking, it is conservative to assume that where algal mats are thick and widespread, they will make an important contribution to wetland primary productivity.

To provide a general characterization of potential algal productivity, we recommend that the presence and general abundance be noted on at least a quarterly basis. Dominant types of algae should be indicated, using the following categories: Floating macroalgal mats (*Enteromorpha, Ulva*); green epibenthic mats (*Enteromorpha*); blue-green epibenthic mats, and diatom films. There is no easy way to obtain accurate estimates of algal abundance. Cover estimates (classes as for vascular plants) can be made within specified areas, e.g., 1-5 m^2 sampling stations that follow the shape of channels or creeks; square-meter segments of the salt marsh, etc.

The overall primary productivity function. Although primary productivity is a basic ecosystem function, there are major problems and errors in measuring productivity rates and calculating the contributions of different producer components for different wetland areas are great. A further concern is the destructiveness of the sampling. Thus, we recommend that productivity studies be considered in research programs, rather than as monitoring objectives.

Given the unreliability of harvest data for estimating NAPP, we are left with only general remarks concerning the primary productivity of southern California coastal wetlands. Onuf (1987) estimated productivity of phytoplankton, benthic microflora, submerged macrophytes and emergent macrophytes for Mugu Lagoon, but only after offering several precautions about the data (ibid., p. 72): "The main caveat about the productivity estimates for the salt marsh vascular plants is the uncertainty arising from the measurement techniques and calculations. The monthly estimates are imprecise because of the high spatial heterogeneity of the marsh. These errors are propagated in the mathematical manipulations used to generate the annual estimates." His general results for the eastern arm of Mugu Lagoon suggest that the benthic microflora and emergent macrophytes are both principal contributors to wetland productivity, with submerged macrophytes (primarily macroalgae) having the highest per-area rates (but covering a small area) and phytoplankton having the lowest per-area and whole-system productivity. This generalization is consistent with results from the salt marsh at Tijuana Estuary, where benthic algal mats appear to be as important to ecosystem productivity as are the vascular plants (Zedler 1980).



Nest of the light-footed clapper rail *Rallus longirostris levipes*

5. Marsh insects: Pollinators, predators, and prey

Objectives. The insects of southern California are responsible for several important salt marsh functions, including pollination, seed dispersal, aerating soils, controlling herbivorous insects, and providing food for birds, small mammals, and other carnivores.

While many of the plants are wind pollinated, there are several species that rely on insects for pollination and seed production. The endangered salt marsh bird's beak (*Cordylanthus maritimus* ssp. *maritimus*) and the regionally rare goldfields (*Lasthenia glabrata*) are important examples. Members of the Coleoptera, Diptera, Hymenoptera, and Lepidoptera are important in pollination. Pollinators link the upper salt marsh to the adjacent coastal scrub-dominated upland, where alternative nectar-producing plants are found. Thus, a fully functional marsh has nearby transitional and upland habitats that maintain an abundance of pollinators.

The herbivore-control function was shown to be critical at the dredge spoil island in south San Diego Bay. Four years after cordgrass was planted, a native scale insect (*Heliaspis spartina*) reached epidemic densities and severely reduced the aboveground biomass of cordgrass. Kathy Williams (SDSU, unpub. data) determined that the native predators (e.g., the beetle, *Coleomegilla*) were lacking, and an experimental control program involving the release of the predator is now underway.

Burrowing insects are an integral part of the soils of salt pannes and higher intertidal marsh areas. While the role of insects in soil aeration and organic matter incorporation remains unquantified, the great abundance of rove beetle burrows (and occasional tiger beetle burrows) suggests great importance.

These and other functions of the hundreds of coastal wetland insects merit assessment of the insect fauna. We recommend general surveys of selected habitat types to characterize the insect communities along with specific censuses of sensitive species (globose dune beetle, wandering skipper, and tiger beetles).

Community sampling methods. Pan traps are used in nontidal areas to trap crawling insects and many flying insects. The method can be used in tidal areas during neap tides, if traps are revisited before a high tide inundates them. Replicate traps (6-10 per site) are randomly placed within areas of relatively uniform plant canopy. Sampling in the warmer parts of the year produces more insects and different species than sampling in cool seasons, according to K. Williams and K. Johnson (SDSU, pers. comm.). At Tijuana Estuary, September samples had abundant beetles and ants, while January samples were dominated by flies. Early spring sampling is suitable for many butterfly species; pollinators would be best sampled in May; late summer is appropriate for tiger beetles.

The pan traps are small cake pans (approx. 23x33x5 cm deep) painted yellow on the inside and placed in a shallow excavation under the plant canopy, so that the edge of the pan is flush with the ground and vegetation. Enough propylene glycol (anti-freeze) is poured into each pan so that the bottom is covered to a depth of about 1 cm. Insects fall into the pans and cannot escape. After about 48 hours, the traps are strained through a sieve and the insects are collected in 70% ethanol in labeled jars. These can be stored for later sorting and identification.

Sweep sampling is necessary to characterize insects in tidal areas and to capture more of the flying insects. Standard butterfly nets are used for this sampling. Uniform numbers of sweeps through vegetation are made along transects of similar length. To sample a marsh area dominated by *Spartina foliosa*, Williams et al. (unpub. report) used 10 transects of 20 m length to sample, with 30 sweeps made along each transect. Transects were parallel and 10 m apart.

Visual surveys are suitable for tiger beetles in mudflat, sandy beach, and salt panne habitats. The numbers seen while walking transects of known length are recorded. Dune beetles may need to be seived from sand near and under dune vegetation. Chris Nagano (US FWS Endangered Species Office, Sacramento) indicates that mark-recapture techniques can be used with these species. His advice should be sought for work with all rare and sensitive insects.

Neither the pan trap nor the sweep methods provide information that can be used to estimate densities, because of the potential for pan or net avoidance. The counts of insects are best interpreted as relative densities for the habitats and times sampled. Such data are valuable indicators of general abundance; information on the abundance or absence of certain functional groups (e.g., pollinators, predators) and of sensitive species is essential for characterizing wetland ecosystems.

Species identification is the biggest problem with characterizing the insect community, and it may not be possible to identify many taxa beyond the family level. However, this is often very useful for examining functional groups. Even general information on size and habit (flying or crawling) will be helpful in characterizing insects as potential food items for consumers such as Belding's Savannah sparrows. Species of special concern, such as the tiger beetles, globose dune beetle, and wandering skipper, need to be identified to species.

Reference data on coastal insects (Williams et al. 1989) have been obtained at Tijuana Estuary from areas of cordgrass (*Spartina foliosa*), tidal and nontidal pickleweed (*Salicornia virginica*), dune, and transition to upland (dominated by flat-topped buckwheat, *Eriogonum fasciculatum*). Six orders of insects comprised the bulk of the insects sampled in September, January, and April: Coleoptera, Collembola, Diptera, Homoptera, Hymenoptera, and Lepidoptera (Table 5.1). In all, 22 orders of arthropods were identified.

Patterns of spatial and temporal variability were found, with upland and transition habitats having the largest numbers of orders (10-11) present and winter generally having the fewest orders represented ($\leq$7, except for the transition habitat, which had 11 orders). Overall, the habitats dominated by cordgrass and pickleweed had the fewest orders (2-8) represented. The summaries of arthropod numbers (Figure 5.1-5.3) indicate substantial differences between habitat types and suggest interactions with season. Ants were primarily in the upland, in September; leafhoppers and planthoppers were most abundant in *Spartina* in January, but not in April; and flies were particularly numerous in *Salicornia*.

Williams and several students are continuing to investigate patterns of insect abundance and distribution at local wetlands, and a better understanding of both structural and functional aspects of the arthropod community should be available in the next few years. Additional species lists, but no quantitative data, are available for Ballona Wetland, north of Los Angeles airport (Nagano, in Schreiber 1981).

Table 5.1. Orders of Arthropoda at Tijuana Estuary, in order of decreasing abundance in pan traps (from Williams, et al., unpub. report). In all, 17,627 individuals were captured.

Order	Common name	% of total
Diptera	flies	28.1
Hymenoptera	bees, wasps	10.4
	Formicidae (ants)	21.9
Homoptera	leafhoppers, planthoppers	19.7
Coleoptera	beetles	6.4
Lepidoptera	butterflies, moths	1.6
Collembola	springtails	1.1
Araneida	spiders	0.8
Isopods	pillbugs	0.6
Amphipods	amphipods	0.5
Dermaptera	earwigs	0.4
Heteroptera	true bugs	0.4
Orthoptera	grasshoppers, crickets	0.4
Acarina	mites, ticks	0.3
Solpugida	wind scorpions	0.1
Thysanoptera	thrips	0.1
Tricoptera	caddisflies	0.1
Odonata	dragon/damselflies	<0.1
Thysanura	silverfish	<0.1
Isoptera	termite	<0.1
Neuroptera	lacewing	<0.1
Phalangida	harvestmen	<0.1
Psocoptera	psocids, book lice	<0.1

Figure 5.1. Relative abundance of selected orders among five habitats sampled at Tijuana Estuary during September 1988 (from Williams et al., unpub. report).

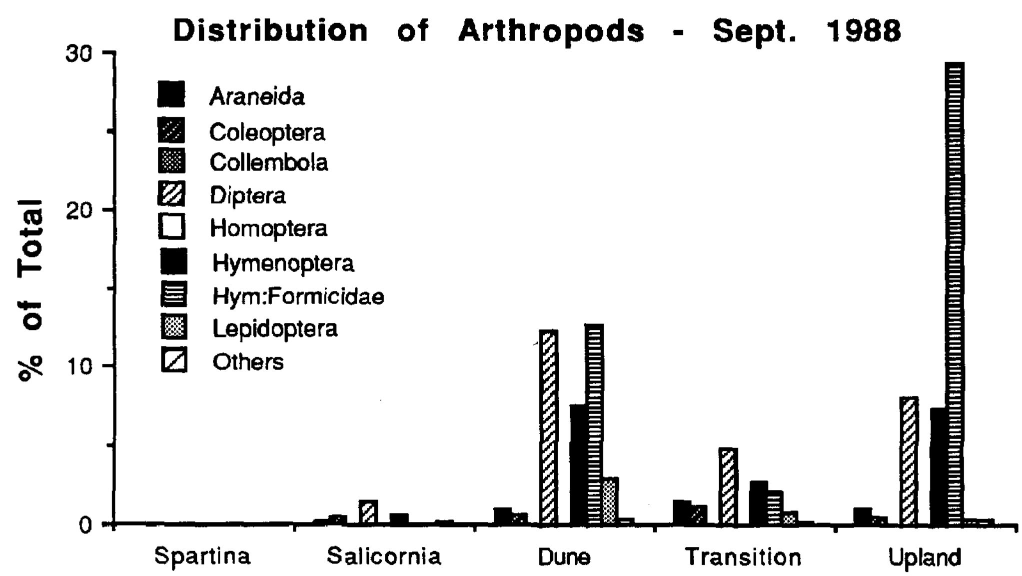


Figure 5.2. Relative abundance of selected orders among five habitats sampled at Tijuana Estuary during January 1989 (from Williams et al., unpub. report).

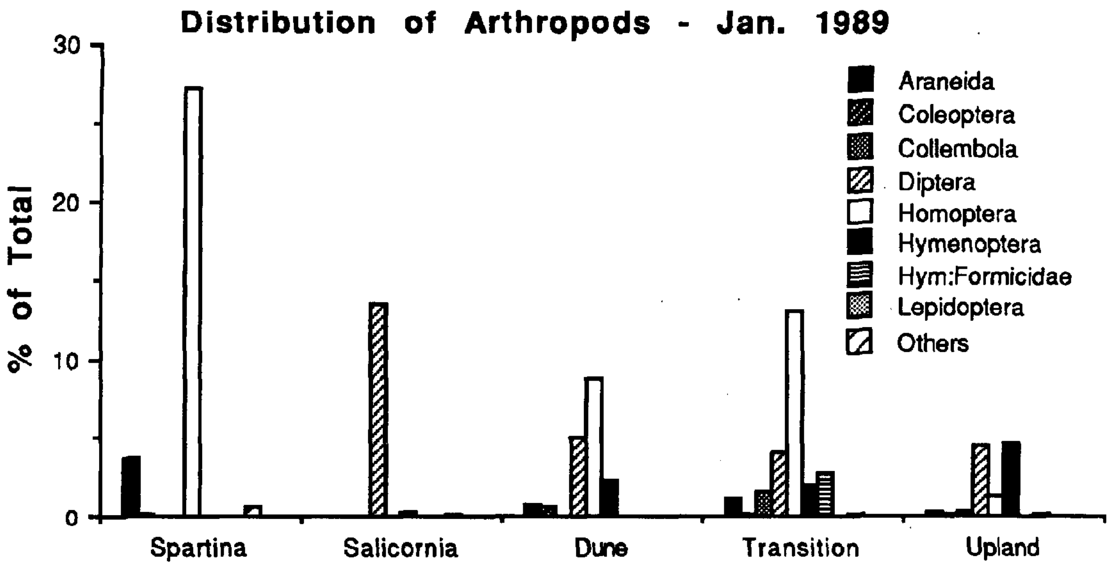
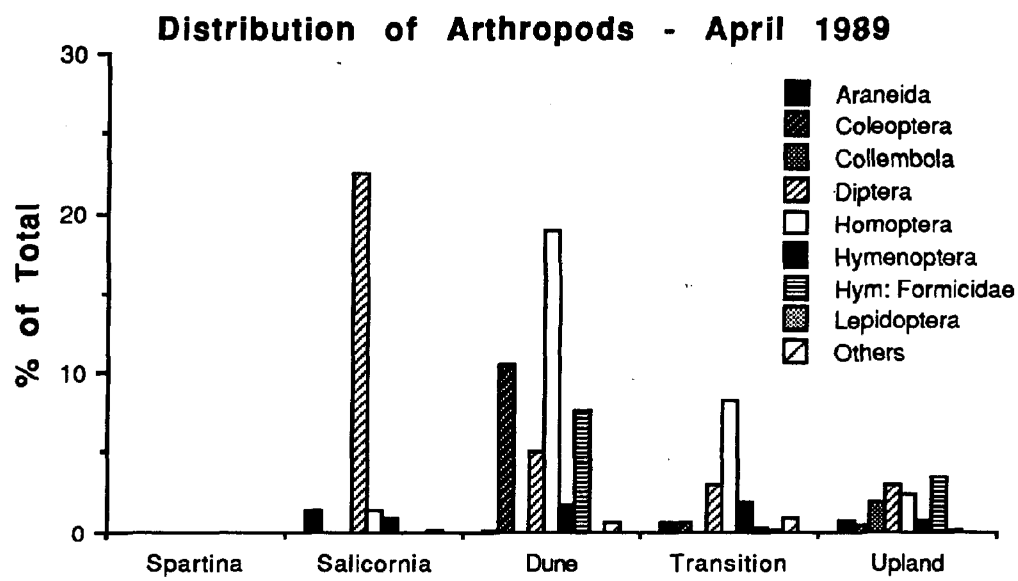


Figure 5.3. Relative abundance of selected Arthropod orders among five habitats sampled at Tijuana Estuary during April 1989 (from Williams et al., unpub. report).



6. Aquatic invertebrates: Food chain support

Objectives. Benthic macroinvertebrates are good indicators of habitat quality and food chain support. Fully tidal wetlands support a large number of benthic invertebrate species, many of which have long life spans and grow to large size. In areas subjected to frequent disturbances, such as dredging and excess freshwater inflows, bivalves may persist, but the species composition will shift from longer-lived to annual species. In the absence of good tidal flushing, clams and ghost shrimp become replaced by species of polychaetes that are tolerant of stagnant water and other environmental extremes. Polychaetes also invade rapidly following catastrophic events, such as rapid salinity reduction (Nordby, SDSU, pers. comm.).

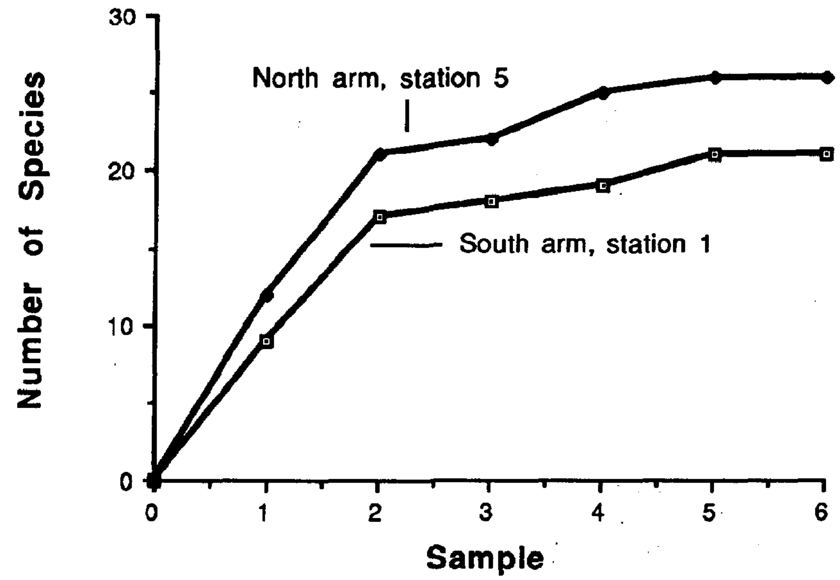
Methods. Two methods are recommended for sampling aquatic invertebrates: sediment cores to assess benthic burrowing forms and litterbag traps for mobile invertebrates on the marsh surface (small crabs, amphipods, and insect larvae). In addition, direct counts of snails and of crab burrows are sometimes used on mudflats. Each is discussed with reference data below. Benthic macroinvertebrates sampling stations are best located near the fish sampling sites (see below), where channel morphometry (width, depth, bank characteristics) is described.

Sediment cores. Benthic macroinvertebrates are collected using a stainless steel cylindrical "clam gun" 45 cm in length and 15 cm in diameter. This device is pressed into the sediment to a depth of 20 cm. Replicate cores are taken within a sampling station. A convenient size station is a circular area of about 2 m in diameter. Usually, five cores per station will include all of the species present (Figure 6.1). The number of samples needed per station varies; more samples are needed to characterize areas that are highly heterogeneous. A species-area curve (cumulative number of species encountered, plotted against cumulative area sampled, cf. Figure 6.1) for each station will indicate whether or not the first few samples have encountered most of the species present.

The method proposed will capture clams and ghost shrimp; worms and some insects will also be obtained. Sieve size is the most important factor to keep constant, and a 1-mm screen is required to compare results with reference data sets. Samples are seived and all obvious animals identified in the field and released. Less obvious animals are taken to the laboratory for identification. Polychaetes are very difficult to identify, but representatives can be preserved for future reference. The numbers of individuals by species should be compared temporally and between stations. Additional information on size of clams for common species is very useful; even the simple measure of the largest individual of each species at each station or notes concerning the relative abundance of large and small individuals would help to document large changes in population structure. Further details (size-frequency distributions) would help describe the food base.

Reference data for benthic cores. Our data from Tijuana Estuary (Table 6.1-6.3) are useful for describing a community under disturbance stress, rather than a restoration target. Historical data from Tijuana Estuary and Mugu Lagoon are used to indicate the "healthy" benthic community that should be expected for sites without dredging, sewage inflows, excessive street drainage, or wastewater discharges.

Figure 6.1. Cumulative number of benthic invertebrate species for stations in the southern and northern arms of Tijuana Estuary. Each sample was the sediment collected with a clam gun and then sieved with a 1-mm screen.



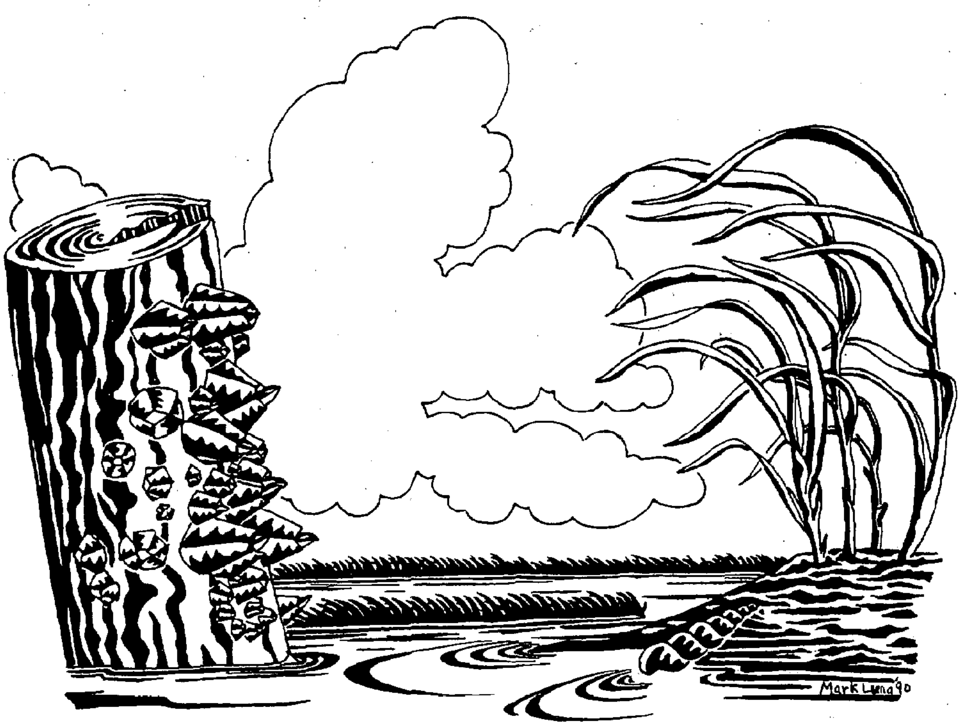


Table 6.1 Mean number of macroinvertebrate species/core sample in the north and south channels of Tijuana Estuary (and s.e.). Data for all species, and for polychaetes and bivalves separately. Capitellid spp. were not identified to species and are not included in the polychaete data. Data are from 6 "clam gun" cores per station; each core had a diameter of 15 cm (area 177 cm^2) and was 20 cm in depth.

	Station	North	South
All macroinvertebrates	1 (near mouth)	15.17 (1.1)	10.83 (0.9)
	3 (upstream of 1)	4.17 (0.4)	3.67 (0.56)
	5 (upstream of 3)	10.83 (0.65)	1.67 (0.5)
	7 (upstream of 5)	3.00 (0.37)	2.3 (0.42)
No. of **Polychaete** spp.	1	6.17 (0.79)	6.17 (0.47)
	3	3.00 (0.26)	0.67 (0.33)
	5	2.83 (0.60)	0.67 (0.33)
	7	1.50 (0.56)	0.00
No. of **Bivalve** spp.	1	3.67 (0.61)	2.33 (0.56)
	3	0.50 (0.22)	1.83 (0.40)
	5	3.33 (2.45)	0.33 (0.21)
	7	0.67 (0.33)	0.17 (0.17)

Table 6.2. Densities of macroinvertebrates in the north and south channels of Tijuana Estuary in November 1988. Data are mean numbers per core sample (n = 6 cores/station) and standard errors (s.e.). Stations are as in Table 6.1.

	Station	North	South
Capitellid worms	1	2.5 (0.67)	114.7 (2.11)
	3	0.17 (0.41)	1.5 (0.81)
	5	8.83 (2.61)	0.00
	7	0.33 (0.33)	0.00
Other Polychaetes	1	19.00 (5.52)	13.17 (1.22)
	3	6.83 (2.45)	1.00 (0.37)
	5	5.17 (1.74)	1.33 (2.45)
	7	5.67 (2.80)	0.00
Bivalves	1	10.17 (1.35)	3.67 (1.05)
	3	0.67 (0.33)	3.5 (0.62)
	5	8.33 (0.99)	0.33 (0.21)
	7	0.83 (0.4)	0.17 (0.17)

Table 6.3. Comparison of numbers of individuals of benthic invertebrates collected at Tijuana Estuary (tidal conditions, but with sewage inflows) and Los Peñasquitos Lagoon (primarily nontidal). Taxa comprising less than 5% are listed as "others." Note that sampling efforts differed for the two wetlands. Data are from Nordby and Zedler (in press).

Taxon	Tijuana Estuary 86-88	Peñasquitos Lagoon 87-88
Sipunculid worms		
Themiste sp.	17	
Echinoderms		
Dendraster excentricus	6	3
Nemertean worms	93	3
Polychaete worms		
Capitellidae	814	521
Spionidae		
Boccardia spp.	68 (5 spp)	183 (4 spp)
Polydora cornuta	267	110
Polydora spp.	63 (2 spp)	210 (2 spp)
Spiophanes missionensis	117	
Unidentified spionid		163
Other taxa combined	437	181
Total polychaetes collected	**1698**	**1368**
Total families collected	**13**	**11**
Total polychaete species collected	**35**	**20**
Bivalve molluscs		
Tagelus californianus	797	40
Protothaca staminea	554	4
Macoma nasuta	221	6
Laevicardium substriatum	30	8
Spisula sp.		17
Other species combined	227	20
Total bivalves collected	**1799**	**95**
Total bivalve species collected	**18**	**12**
Decapod crustaceans		
Callianassa californiensis	234	3
Phoronida		
Phoronis sp.	1	114
Brachiopoda		
Glottida albidia	____	1
Total sampling area (cumulative area in m^2)		
bivalves	4.77 m^2	3.34 m^2
other taxa	3.34 m^2	3.34 m^2
Total number of quarterly samples		
bivalves	10	7
other taxa	7	7

Macroinvertebrates in the north and south main channels of Tijuana Estuary illustrate differences that relate to tidal flushing (Tables 6.1-6.2). In November 1988, the north channel (Oneonta Slough) was fully tidal, while the south channel had very little tidal influence, except at the station nearest the mouth. Mouth stations of both channels were somewhat influenced by sewage inflows from the Tijuana River.

Data from long-term sampling of the northern arm of Tijuana Estuary (Table 6.3) indicate that even though the estuary is stressed by wastewater inflows, the benthic community is still more diverse than that of Los Peñasquitos Lagoon, which is primarily nontidal.

Historic data from Tijuana Estuary, before major floods and sewage inflows, serve to characterize a "healthy" benthic community. Peterson (1972, 1975, 1977) sampled both Tijuana Estuary and Mugu Lagoon (34°N, 119°W) in the 1970's. His data for live bivalves in saline habitats show that *Nuttallia (Sanguinolaria) nuttallii* and *Protothaca staminea* were the most abundant bivalves at both study sites. *Tagelus californianus*, *Cryptomya californica*, *Macoma nasuta*, and *Laevicardium substriatum* were also present but in lower numbers.

Hosmer (1977) sampled bivalve composition and measured the sizes of representatives of each species. Large individuals were abundant. The mean sizes for the dominant bivalves were: 71 mm for *Nuttallia nuttallii* ; 22 mm for *Protothaca staminea* ; and, 27 mm for *Tagelus californianus*. His results contrast with recent data, i.e., *Nuttallia nuttallii* no longer exists at Tijuana Estuary, and *P. staminea* is, on the average, half as large.

Litterbag traps

We have found litterbag traps to be very useful in collecting small invertebrates from marsh habitats. The placement of such traps in the wetland for a month-long period allows a variety of small invertebrates (flies, amphipods, beach hoppers, crabs, and snails) to colonize the litter and be collected easily without damage to the habitat. However, the animals trapped are an indicator of what is available to colonize substrates, and are not necessarily representative of densities of organisms living outside the traps. Thus, the method provides an index of this invertebrate community, which is especially useful for simultaneous comparisons of natural and man-made wetlands.

Stations in the upper and lower intertidal marsh will attract different species and densities of organisms. The method was modifed from Levings' (1976) basket traps. Litterbags are made from 1-cm nylon mesh (black is preferred). Dried plant material weighing 40 g is placed inside each bag and the ends of the bag are folded over and stapled. The resulting bag is about 15 x 25 cm. For litter as filler, Rutherford (1989) used dried cordgrass *(Spartina foliosa)* left over from harvests of experimental material at PERL. In comparisons using native cordgrass and commercial straw, it was clear that straw trapped fewer animals. Comparisons between native cordgrass and a mixture of cattails and bulrushes gave similar results (Table 6.4). Because cordgrass should not be taken from native stands, we recommend the use of cattails and bulrushes from man-made wetlands.

Three replicate litterbags should be placed at each sampling station. Stations of about 2 m in diameter are convenient. Bags are secured at the soil surface with wire flagging stakes. Seasonal sampling produces different dominants, with the largest numbers of individuals occurring in winter. Litterbags should be left in the marsh for one month, then collected and placed in zip-lock storage bags containing a 10% buffered formalin seawater solution. Litterbags may be stored in formalin for up to 5 days before gently washing the litter over a 0.5 mm sieve to collect the organisms. The catch is preserved in 70% isopropyl alcohol, and identification requires microscopic (dissecting scope) examination.

Reference data for litterbag traps. Data from Tijuana Estuary, Sweetwater Marsh, and Paradise Creek all indicate that sampling time and location are important variables. No set of data can be recommended as a reference point with which to compare a constructed wetland. Rather, simultaneous sampling in natural and man-made marshes is recommended. At the Sweetwater River Wetland Complex, a comparison of natural and constructed habitats (Rutherford 1989) shows that about 3 times as many individuals and consistently more species were collected in the natural marsh than in the mitigation marsh, which was assessed 4 years after construction.

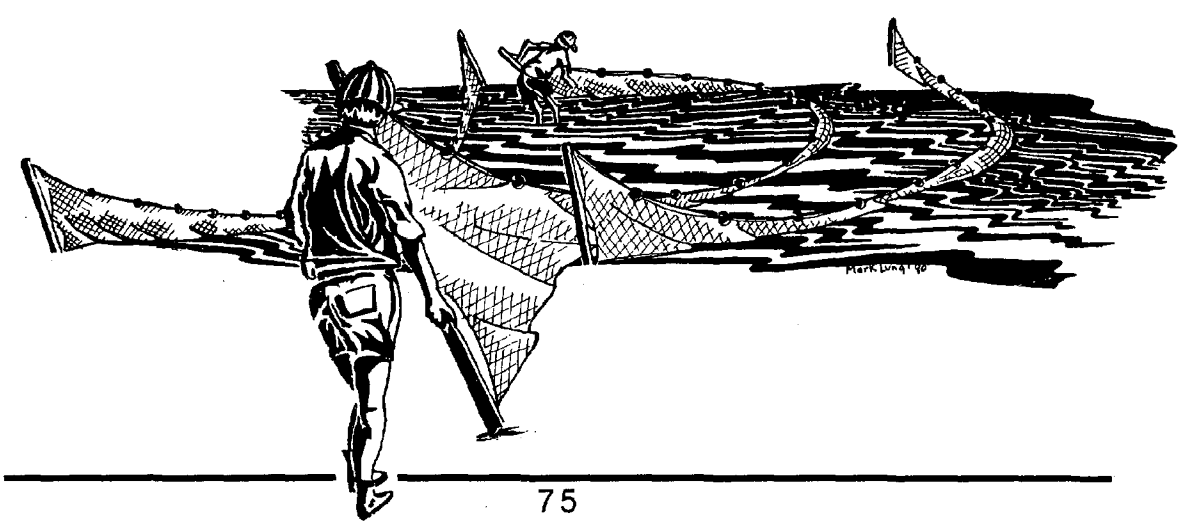
Results from bags collected in January 1989, after one month in the Tijuana Estuary, are provided to indicate the kinds of species collected in litterbag traps. The data are from the lower marsh and adjacent mudflats. Only the results for bags made using *Spartina* are presented.

Table 6.4. Relative abundance (% of total) of invertebrates trapped in litter bags that were filled with dried *Spartina*, *Typha*, or *Scirpus*. Litter bags were placed in Paradise Creek Marsh for the month of March 1989. Each site had 4 replicate bags. The % similarity of invertebrate composition (based on abundance) comparing *Spartina* and *Typha* litter bags was 78%, comparing *Typha* and *Scirpus* was 75%, and comparing *Scirpus* and *Spartina* was 86%.

	Spartina	*Typha*	*Scirpus*
Pericoma spp. (fly larva)	78.4	58.4	83.1
Collembola (springtails)	5.8	6.2	<1
Tromboidea (???)	5.8	4.5	2.4
Assiminea californica (small snail)	3.9	5.1	1.3
Unknown larvae	1.2	8.0	4.7
Capitellid worms (polychaete)	1.6	6.6	5.1
Orchestia traskiana (amphipod)	<1	7.6	2.0
Ligia occidentalis (isopod)	<1	2.4	<1
Saldidae (fly)	1.8	<1	<1
Other	<1	<1	<1

Table 6.5. Invertebrates collected at three stations, 2 intertidal elevations per stations, in the north channel (Oneonta Slough) of Tijuana Estuary. Station 1 (S1) was near the ocean inlet; station 7 was a tidal creek near the northern limit of the wetland. *Spartina* was used as filler for all bags. Four taxa with only 1 individual in n=18 bags were included in the total but not listed.

Organism	S1 lower	S1 upper	S6 lower	S6 upper	S7 lower	S7 upper	Mean	s.e.
Capitellidae	6	0	2	0	5	9	3.7	1.5
Spionidae	61	0	0	0	0	1	10.3	10.1
Polydora ligni	0	0	0	0	7	0	1.2	1.2
Streblospio benedicti	0	0	0	0	2	0	0.3	0.3
Assiminea californica	0	27	0	206	2	72	51.2	33.0
Melampus olivaceus	0	0	0	0	0	4	0.7	0.7
Hemigrapsus oregonensis	26	0	9	0	32	10	12.8	5.5
Pachigrapsis crassipes	23	14	19	5	3	0	10.7	3.8
Juvenile crab	11	0	8	2	7	5	5.5	1.6
Orchestia traskiana	4	17	4	6	12	45	14.7	6.4
Other amphipods	0	0	0	0	12	0	2.0	2.0
Ligia occidentalis	0	18	0	0	22	76	19.3	12.1
Diptera	2	45	0	1	0	0	8.0	7.4
Dipteran larvae	1	2	0	5	5	35	8.0	5.5
Dipteran larvae B	0	0	0	0	3	0	0.5	0.5
Pupae	3	25	0	3	0	2	5.5	3.9
Unknown larvae	0	1	0	1	0	0	0.3	0.2
Unknown insect	0	0	0	3	0	3	1.0	0.6
Totals =	**137**	**149**	**42**	**235**	**112**	**263**	**156.3**	**33.2**



7. Fishes: Community dynamics, controlling factors

Objectives. Fishes are valuable indicators of ecosystem health. Generally, the presence of few species (low species richness) indicates stressful environmental conditions. Gobies and mullet are relatively tolerant of environmental extremes and are among the last species to disappear. Fishes are valued as foods for birds that use the estuary. A few species are of recreational or commercial interest, e.g., California halibut and longjaw mudsuckers (as baitfish).

Methods. Fishes are sampled quarterly (March, June, September, and December) at replicate stations. The habitat types to be sampled for fishes include deep channels, tidal creeks, deep saline ponds, brackish creeks, and freshwater ponds. Fish sampling is carried out at moderate tide levels, when stations are accessible by foot. Sampling at times of moderate tidal amplitude also avoids tidal transport of transient species.

Blocking nets prevent fish from escaping the sampling area. Standard mesh size allows comparisons among seining efforts. Adult and juvenile fishes should be collected using 3-mm mesh blocking nets and bag seine. The 3-mm size mesh ensure the capture of small yet ecologically important goby species. A linear distance (e.g., 10-15 m) parallel to the creek or channel sampled should be measured and the channel nets deployed to confine all fishes within the two nets. The bag seine is then drawn in a circle within the blocking nets and pulled to shore. For pond habitats, a semicircular area should be enclosed with a net seine, and the seine drawn toward the shore. The species composition and numbers collected per tow are recorded, a subsample measured (for length-frequency distributions), and fishes released outside the blocking nets. Repeated tows must be made and continued until the total number of fishes collected declines in two successive seinings. The first and second seinings capture the water-column species, while later seinings collect bottom-dwelling species. Stomping on the bottom scares up benthic species (e.g., burrowing gobies) that would otherwise allow the net to drag over them. The blocking nets are then closed by drawing the two together in a semi-circle and then pulling them to shore, thereby capturing those fishes that managed to escape the bag seine. The successive seining results can be used to estimate total population size, using the standard catch-per-unit-effort technique, plotting number per catch against cumulative catch and extrapolating the line to obtain the cumulative catch when catch is zero. The area seined (length of segment and channel width) should be measured so that densities can be calculated on a per-surface-area basis. Maximum depths are recorded.

Reference data comparing two fish sampling techniques. To demonstrate the need for repeated seining to obtain representative data for fish densities and species composition, a fish sampling exercise was carried out by Chris Nordby at the "east-west channel" at Tijuana Estuary on March 11, 1987. A 10-m segment of the channel was blocked at each end with seines of 3-mm mesh, and the blocked area was sampled with a third 3-mm seine in four repeated efforts. Each of the four seining efforts was equal in area and seine length. Then, the two blocking nets were "closed" by pulling each toward the center. The combined fish catch in this fifth seining effort was a double effort, so the numbers of fish caught were halved to compare with the first 4 seining efforts. Water depth was approximately 1 m on the outgoing tide.

Using the catch-per unit effort method (Figure 7.1), it was determined that the total fish population within the blocked portion of the channel was 1470 fish ($Y=159 -0.108X$; for $Y=0$, $X=1470$; i.e., when no more fish would have been present, 1470 would have been caught). This catch-per-unit-effort calculation provides a good estimate of total population when sequential catches decline smoothly, as in this examnple. However, in many cases, a catch with few fish can be followed by one with many fish, as gobies move into the water column. We recommend seining sequentially until the catch is clearly diminished, then summing the numbers caught to estimate the number of fish per area.

In the test case (Table 7.1), the first seining yielded 157 fish; which was only about 10% of the estimated fish that could have been caught there. For the 5 comparable seining efforts, a total of 643 fish were recorded. Of these, 395 were gobies, 160 topsmelt, 64 sculpin, 11 killifish, and 13 flatfish.

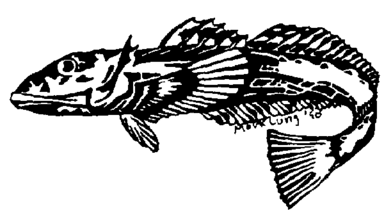
Table 7.1. Comparison of single and repeated seining efforts at Tijuana Estuary. Data are relative abundances, obtained by C. Nordby, SDSU, on March 11, 1987. Species that appeared to be dominant are shown in bold type.

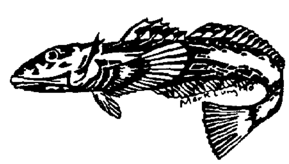
	1st seining effort	Total in 5 seinings
Topsmelt	**70.1%**	24.9%
Gobies	6.4	**61.4**
Sculpin	17.8	10.0
Killifish	4.5	1.7
Flatfish	1.3	2.0

The first seining suggested that topsmelt were dominant; however, as repeated seinings show, they are merely the most catchable species. In fact, 69% of the topsmelt found were caught on the first seining. Topsmelt are water-column species that do not readily avoid seines.

Additional sampling showed that gobies were the dominant members of the fish community. Gobies are bottom-dwelling species that respond to the repeated disturbances of seining; their numbers increased progressively in the first three seinings (Figure 7.2), and declined slightly in the last two efforts. Two mullet were caught in the later seinings, but were not included in these data summaries; it is well known that seines undersample mullet because they swim or jump out of the seines.

We conclude from this comparison of seining methods that: A single seining can underestimate fish density by as much as an order of magnitude. A single seining can misrepresent the fish community; the most catchable species will appear to be the most abundant ones, while the least catchable species will be absent or undersampled. In this data set, repeated sampling produced only one additional species (mullet), one that is generally undersampled due to its net avoidance behavior.





Staghorn sculpin
(Leptocottus armatus)

Figure 7.1. Repeated seining of a blocked channel plotted against the prior cumulative catch. Data of C. Nordby, PERL.

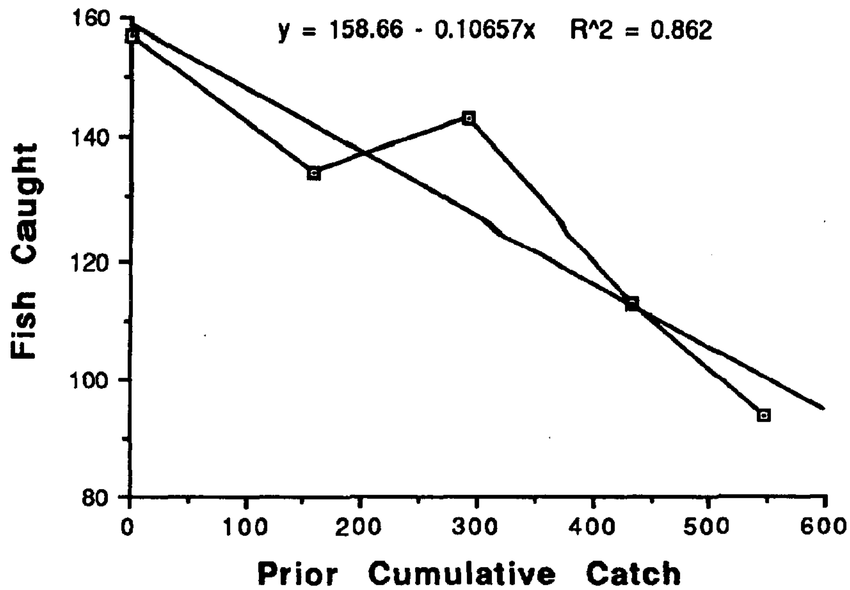
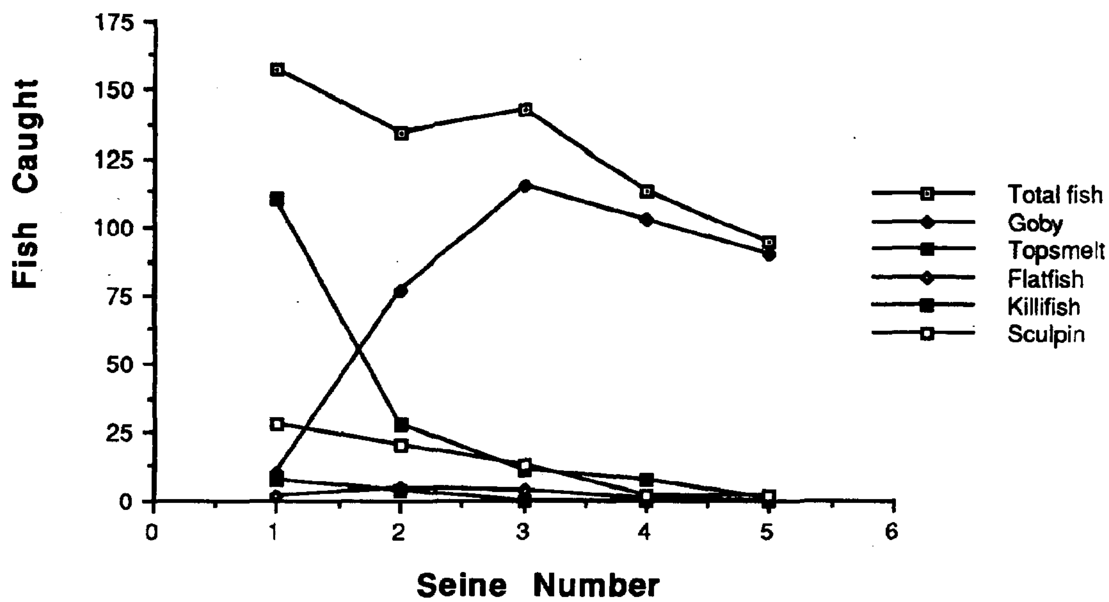


Figure 7.2. Species composition of repeated seinings. Data of C. Nordby, PERL.



Reference data for fishes of wetland channels. Nordby has used the same sampling methods to assess populations of fishes at Tijuana Estuary and Los Peñasquitos Lagoon for several years (Table 7.2, Nordby and Zedler in press). The sampling program indicates large differences in species composition and density for the two wetlands and for stations within Tijuana Estuary. Differences are related to the history of hydrologic disturbances at the two wetlands, with Tijuana Estuary being open to tidal flushing at all times during the sampling and Los Peñasquitos Lagoon closing annually for much of the warm season. During the closures, there were floods in both October 1987 and December 1988, the fresh water was impounded, and large numbers of fish were killed after salinity and oxygen levels dropped. At Tijuana Estuary, raw sewage inflows were persistent each year, and the sampling station closest to the Tijuana River was most depleted of its fish community.

Both types of hydrologic modifications are human impacts--the sewage flows from Mexico have changed the river from an intermittent stream to a year-round, nutrient-rich river that is laden with toxic materials. The prolonged closure to tidal flushing has resulted from the construction of roads and a railroad (and associated filling and reduced tidal prism). Thus, none of the fish communities we describe serves as a reference datum for a "healthy" tidal wetland. However, the differences between the two systems have helped to show patterns of decline that follow hydrologic disturbances. Low numbers of species and single-species dominance indicate low-quality habitat; absence of longjaw mudsuckers may indicate a recent period of hypersalinity; the presence of mosquitofish indicates high or persistent freshwater inflows.

The total species lists (Table 7.3) for Tijuana Estuary and Los Peñasquitos Lagoon indicates fewer species in the latter wetland, but note that the sampling effort was also less.

Ichthyoplankton sampling. If juvenile or adult fishes are not found using seines, the habitat might still be suitable but larvae may not be available for settling. In this case, ichthyoplankton sampling should be considered to determine if young are available for colonization. The lack of ichthyoplankton would indicate that a basic ecosystem function is missing. Two periods are recommended for sampling ichthyoplankton--sampling in March will capture nearshore species that move into the estuary with tidal waters, and sampling in April will assess availability of larvae of resident species such as topsmelt and gobies. Nordby (1982) gives detailed information on the year-round distribution of ichthyoplankton at Tijuana Estuary.

Table 7.2. Fishes in two southern California coastal wetlands (from Nordby and Zedler in press). Data are relative abundances (% of total caught in multiple censuses each year).

	Tijuana Estuary			Peñasquitos Lagoon	
Species	1986-87	1987-88	1988-89	1987-88	1988-89
Atherinops affinis	52	14	7	38	36
Clevelandia ios	41	58	90	22	14
Fundulus parvipinnis	4	19	1	4	2
Gillichthys mirabilis		<1	<1	28	14
Gambusia affinis				<1	24
Others	3	~8	~1	~7	10

Table 7.3. Fish species collected at Tijuana Estuary and Los Peñasquitos Lagoon (modified from Nordby and Zedler in press). X = less than 1% of annual catch.

Taxon		Common name	Tijuana Estuary 1986-88	Peñasquitos Lagoon 1987-88
Atherinidae	*Atherinops affinis*	topsmelt	15,437	1,875
Blennidae	*Hypsoblennius gentilis*	bay blenny	X	
	Hypsoblennius gilberti	rockpool blenny	X	
	Hypsoblennius jenkinsi	mussel blenny	X	
Bothidae	*Paralichthys californicus*	California halibut	X	X
Cottidae	*Leptocottus armatus*	staghorn sculpin	1,431	346
	Artedius sp.	sculpin	X	
Cyprinodontidae	*Fundulus parvipinnis*	California killifish	2,367	107
Engraulidae	*Anchoa compressa*	deepbody anchovy	X	67
Girellidae	*Girella nigricans*	opaleye	X	X
Gobiidae	*Clevelandia ios*	arrow goby	60,097	816
	Gillichthys mirabilis	longjaw mudsucker	275	877
	Ilypnus gilberti	cheekspot goby	X	X
	Lepidogobius lepidus	bay goby		X
	Quietula y-cauda	shadow goby	X	
Mugilidae	*Mugil cephalus*	striped mullet	X	X
Plueronectidae	*Hypsopsetta guttulata*	diamond turbot	X	X
	Plueronichthys ritteri	spotted turbot	X	
Poeciliidae	*Gambusia affinis*	mosquitofish		937
Rhinobatidae	*Rhinobatus productus*	shovelnose guitarfish	X	
Serranidae	*Paralabrax clathratus*	kelp bass	X	
Sciaenidae	*Seriphus politus*	queenfish	X	
Syngnathidae	*Syngnathus leptorynchus*	bay pipefish	X	X
Total fishes collected			**80,165**	**5,087**
Total species encountered			**21**	**13**
Total sampling effort (cumulative area in m^2)			**4,795**	**1,985**
Number of quarterly samples			**12**	**8**

8. Birds

Objectives. The most obvious and widely appreciated animals of the coastal wetlands are the birds. In recent years, several studies have quantified coastal wetland birds (Boland 1981, 1988) and identified their functional roles (Boland 1988, Quammen 1984). Additional work on endangered species has been done, and some sites are monitored yearly for target species (US FWS). However, long-term monitoring programs of the total bird community are still needed. Because birds are a major linkage between wetlands, coordinated monitoring programs of all water-associated birds throughout the Pacific flyway are needed. In addition, we need to determine how different habitats attract birds and the length of time it takes target species to find and use constructed wetlands.

Bird community sampling. Several methods have been developed for sampling terrestrial birds (Ralph and Scott 1981), which pose special problems because of their high mobility and low visibility in areas of dense vegetation. In coastal wetlands, many of the species of interest are fairly large, occur in groups, and feed in the open. Flocks of shorebirds are highly visible. Still, there are difficulties in selecting consistent habitat areas, since tides expose different widths of shoreline each day. Observers need to be skilled in estimating distances, since counts of birds per area could easily be biased when transects of unmeasured width are censused. Walking transects along shorelines allows counts to be expressed as number per length of transect, even if transect width varies. However, data from transects of fixed width and length are easiest to analyze. Boland (1988) used multiple transects of fixed length (50 m) to obtain densities (means and standard errors) for each habitat.

Habitat types are first delineated, and transects chosen to represent each area. Aerial photos should be used to select areas of relatively homogeneous vegetation and topography. Several short transects should prove more useful than a few long transects in comparing mean densities among habitat types, as the standard errors are likely to be lower (Hanowski et al. 1990). Census dates and times are then selected, keeping in mind that coastal bird communities change with season, with inclement weather, and with tidal condition. Patrice Ashfield and Barbara Kus (see below) sample weekly during most of the year (biweekly in summer) in order to compare bird communities at high tide and low tide, and in order to follow changes with migration.

Reference data. A detailed, year-long sampling program was initiated at Tijuana Estuary during 1988-89 (B. Kus and P. Ashfield, SDSU, pers. comm.). The program had weekly censusing of shorebirds and waterfowl (density by species) during months of high activity (Oct.-Dec.) to include weeks with both spring and neap tides. Thereafter (Jan.-June), censuses were biweekly, and only at low tide. In this sampling program, counts were recorded by habitat type within three main transects. The two transects with greatest bird activity were censused simultaneously by qualified observers, to allow comparison under the same tidal conditions. The weekly surveys alternately characterized bird use under low and high tides. Censuses began one hour before the tide condition that was being characterized. Observers carried binoculars, a spotting scope, and a two-way radio to maintain contact with one another and reduce duplicate counting of individuals flying between transects. Counts were made by species, noting the specific habitat type (Table 8.1), inundation level (with or without standing water) and noting behavior within three classes: foraging, roosting, or flying. During each visit, additional notes were made on selected species (raptors, Belding's Savannah sparrows, and light-footed clapper rails) that were seen outside the regular transect.

Table 8.1. Habitat types used in censusing birds at Tijuana Estuary (B. Kus, SDSU, pers. comm.)

Habitat	Subhabitats
Beach	Lower (surf line) and Kelp (wrack)
Dune (sandy beach in estuary)	Distance from channel: within 3 m and >3 m
Unvegetated intertidal	Inlet, Cobbles, Channel
Vegetated intertidal	Distance from channel: 0-5 m, >5 m
Salt panne (bare flats, often dry)	

Table 8.2. Summary data for birds censused along three transects within Tijuana Estuary in 1988-89. Data from Kus and Ashfield (SDSU, unpub. data). Data for biweekly low-tide censuses are for the period of October 7, 1988, through April 8, 1989.

Group	No. of species	% of individuals*
Small waders	14	37.31
Gulls and terns	13	27.84
Large waders	8	18.24
Waterfowl	16	14.45
Herons and egrets	7	0.47
Raptors	9	0.39
Grebes and pelicans	5	0.14
Land birds	18	not included
Total number of species	**90**	
Total sightings in low tide censuses on 14 dates		**25,572**

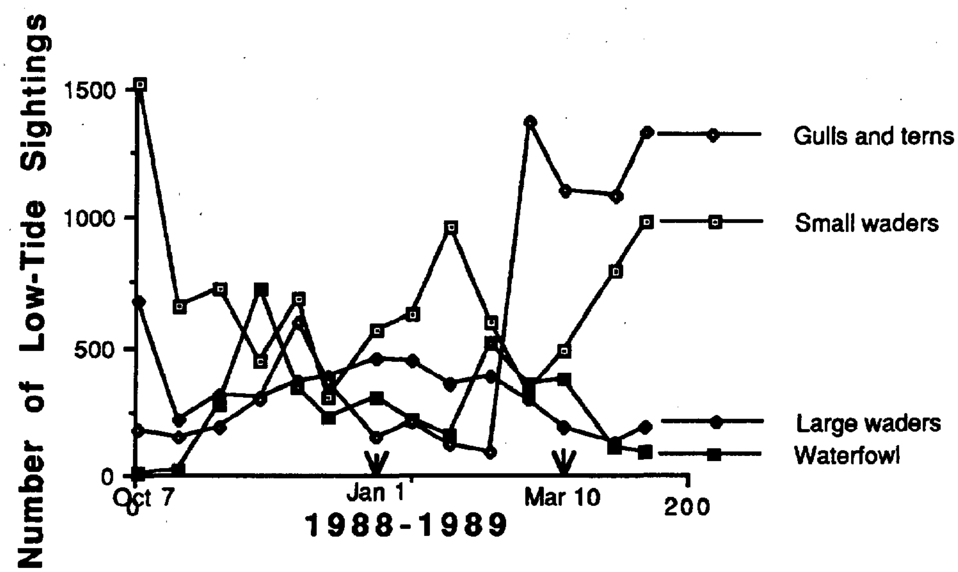
Table 8.3. Percent of total observations in each of 5 habitats during low tide surveys on 14 dates of the northern half (the most well-flushed portion) of Tijuana Estuary. There were 23,298 bird sightings within these groups in the two northern transects.

Habitat	Small waders	Gulls, terns	Large waders	Water-fowl	Herons egrets	Grebes, pelicans
Unvegetated intertidal	77.2	22.3	72.7	98.7	92.9	89.5
Vegetated intertidal	0.5	0.0	2.1	0.7	6.1	0.0
Beach (open coast)	1.3	5.4	2.6	0.0	0.0	0.0
Dune (beach in estuary)	14.8	6.0	17.9	0.6	1.0	0.0
Island	6.2	66.3	4.8	0.1	0.0	10.5
Total sightings	8,221	7,159	4,458	3,343	98	19

Table 8.4. Species list for 7 groups of birds at Tijuana Estuary (Kus and Ashfield, unpub. data). The list of land birds includes only species seen at the three census locations.

Small waders	
Actitis macularia	Spotted sandpiper
Pluvialis squatarola	Black-bellied plover
Charadrius vociferus	Killdeer
C. semipalmatus	Semipalmated plover
C. alexandrinus	Snowy plover
Arenaria interpres	Ruddy turnstone
A. melanocephala	Black turnstone
Calidris canutus	Red knot
C. mauri	Western sandpiper
C. minutilla	Least sandpiper
C. alpina	Dunlin
C. alba	Sanderling
Gallinago gallinago	Common snipe
Rallus longirostris levipes	Light-footed clapper rail
Large waders	
Himantopus mexicanus	Black-necked stilt
Recurvirostra americana	American avocet
Catoptrophorus semipalmatus	Willet
Tringa sp.	Yellowlegs
Numenius phaeopus	Whimbrel
N. americanus	Long-billed curlew
Limosa fedoa	Marbled godwit
Limnodromus sp.	Dowitchers
Waterfowl	
Branta canadensis	Canada goose
Anas americana	American wigeon
A. strepera	Gadwall
A. crecca	Green-winged teal
A. platyrhynchos	Mallard
A. acuta	Northern pintail
A. discors	Blue-winged teal
A. cyanoptera	Cinnamon teal
A. clypeata	Northern shoveler
Aythya affinis	Lesser scaup
Melanitta perspicillata	Surf scoter
Bucephala clangula	Common goldeneye
B. albeola	Bufflehead
Mergus serrator	Red-breasted merganser
Oxyura jamaicensis	Ruddy duck
Fulica americana	American coot
Raptors	
Asio flammeus	Short-eared owl
Pandion haliaetus	Osprey
Elanus caeruleus	Black-shouldered kite
Circus cyaneus	Northern harrier
Accipiter cooperii	Cooper's hawk
Buteo jamaicensis	Red-tailed hawk
Falco sparverius	American kestrel
F. columbarius	Merlin
F. peregrinus	Peregrine falcon
Gulls and terns	
Larus philadelphia	Bonaparte's gull
L. heermanni	Heermann's gull
L. delawarensis	Ring-billed gull
L. canus	Mew gull
L. californicus	California gull
L. argentatus	Herring gull
L. occidentalis	Western gull
Sterna caspia	Caspian tern
S. maxima	Royal tern
S. elegans	Elegant tern
S. hirundo	Common tern
S. forsteri	Forster's tern
Rynchops niger	Black skimmer
Pelicans, grebes, and cormorants	
Podilymbus podiceps	Pied-billed grebe
Podiceps nigricollis	Eared grebe
Aechmophorus occidentalis	Western grebe
Phalacrocorax auritus	Double-crested cormorant
Pelecanus occidentalis	Brown pelican
Herons and egrets	
Ardea herodias	Great blue heron
Casmerodius albus	Great egret
Egretta caerulea	Little blue heron
E. rufescens	Reddish egret
E. thula	Snowy egret
Nycticorax nycticorax	Black-crowned night heron
Phoenicopterus sp.	Flamingo
"Land birds"	
Ceryle alcyon	Belted kingfisher
Sayornis nigricans	Black phoebe
Sayornis saya	Say's phoebe
Eremophila alpestris	Horned lark
Hirundo rustica	Barn swallow
Corvus corax	Common raven
Lanius ludovicianus	Loggerhead shrike
Cistothorus palustris	Marsh wren
Thyromanes bewickii	Bewick's wren
Chamaea fasciata	Wrentit
Anthus spinoletta	Water pipit
Carpodacus mexicanus	House finch
Geothlypis trichas	Common yellow-throat
Passerculus sandwichensis beldingi	Belding's Savannah sparrow
Melospiza melodiaa	Song sparrow
Sturnella neglecta	Western meadowlark
Columba livia	Rock dove
Mimus polyglottos	Northern mockingbird

Figure 8.1. Temporal variability in numbers of sightings for the most common groups of birds using Tijuana Estuary (Kus and Ashfield, unpub. data).



Sweetwater River Wetland Complex (SRWC). For her M.S. thesis, Ashfield (in prep.) censused birds along several permanent transects along San Diego Bay. Weekly sampling for most of the year (biweekly in summer) provided sufficient data to compare bird use during both high and low tides. Some of her results have been summarized and used in evaluating the bird community of the constructed marsh (Section III); the relative abundances for the two natural wetland sites are repeated here (Table 8.5), with rare species included. When completed, Ashfield's thesis should be consulted for details of other sampling locations and seasonal patterns of abundance.

Santa Margarita Estuary. Birds were censused biweekly at SME for one year, beginning April 1986 (Hollis et al. 1988). During late winter/early spring of 1987, the ocean inlet closed, and the reduced abundance and diversity of birds was attributed to reduced tidal flushing. Coots increased and ruddy ducks were higher than the previous year, while other water birds had reduced numbers. Shorebird use was characterized as especially low in recent years. Habitats sampled included willow woodland, river channel, marsh, ponds, salt flat, and sewer ponds. Densities were lowest from May-July and highest from late fall to early spring.

Table 8.5. Water-associated birds at SRWC. Data of Ashfield (in prep.) are for 13 months in 1989-90, for low-tide censuses, for all species (excepting gulls) that made up >0.1% of the sightings. PC = Paradise Creek and BS = Bay Shoreline off Gunpowder Pt.

% of total sightings	PC	BS
Western sandpiper	15.5	40.6
Dowitcher spp.	25.0	11.7
Willet	20.0	7.2
Marbled godwit	4.9	8.5
Red knot	0	8.7
Dunlin	5.7	1.7
Bufflehead	6.6	0
Killdeer	4.3	0.1
Great blue heron	4.0	0.1
Surf scoter	0	3.8
Lesser scaup	0.3	3.3
Western grebe	0	3.1
Least sandpiper	2.1	0.7
Pied-billed grebe	2.3	<.1
Snowy egret	1.2	1.0
Long-billed curlew	1.6	0.5
Brant	0	1.9
Great egret	1.7	0.1
Black-bellied plover	0.2	1.6
Black-necked stilt	1.4	0
Spotted sandpiper	1.2	0
Greater yellowlegs	0.5	0.6
Forster's tern	0.7	0.3
Northern pintail	0	0.9
Semipalmated plover	0	0.8
Ruddy turnstone	0	0.5
Sanderling	0	0.4
Mallard	0.3	0
American widgeon	0	0.3
Northern shoveler	0	0.3
Black-crowned night-heron	0.2	0
Red-breasted merganser	0.2	0
American avocet	0.2	0
Least tern	0	0.2
Black skimmer	0	0.2
Horned grebe	0	0.2
Brown pelican	0	0.1
Green-winged teal	0	0.1
Snowy plover	0	0.1
Whimbrel	0	0.1
Total number of species	**23**	**39**

Other bird counts. Local sources of information on bird use may provide important historical records. The Audubon Christmas Bird Count data from wetland sites should be obtained on an annual basis (published in special editions of *American Birds*). In addition, the Pt. Reyes Bird Observatory (4990 Shoreline Highway, Stinson Beach, CA 94970) has information on shorebirds for selected coastal water bodies. A recent effort of the Observatory was to count shorebirds in all wetlands between Baja California and British Columbia on a single date, April 22, 1989. This is a very useful comparative data base. For San Diego County, the census resulted in over 12,000 shorebird sightings, of which approximately 75% were small waders (over 50% were western sandpipers). The largest concentration of shorebirds was in south San Diego Bay, with the next largest group in the Flood Control Channel of the San Diego River.

Endangered bird populations. Clapper rails are censused during the breeding and nesting season by listening to their calls. Observers stand quietly at a vantage point. Calls are most frequent at dusk, and surveys begin 0.5 hr before dusk and last 0.5-1.0 hr after dark, depending on calling activity. Less calling leads to longer census periods. Experts can often distinguish paired and unpaired birds by their call: a "clatter" call usually indicates a pair especially if the male and female call in a duet; a "kek" call is made by a searching male or female. Unpaired females searching for a mate is a third type of call.

In addition, the presence of rails can be identified using recorded calls (during the territorial season), following the techniques developed by Barbara Massey (Endangered species consultant, Long Beach), Dick Zembal (US FWS, Laguna Niguel), and Paul Jorgensen (Calif. Dept. of Parks and Recreation).

The recovery plan for the light-footed clapper rail (LFCR Recovery Team 1977, p. 15) states the objective of increasing the breeding population to "at least 400 pairs by preserving and restoring approximately 4,000 acres of wetland habitat in at least 15 marshes..." Data for several years (Table 8.6) show high variation in the rail population, with all values well below the target of 400 pairs.

Table 8.6. Historical data for light-footed clapper rails in southern California (R. Zembal and B. Massey, US FWS, unpub. report). Not all 35 wetlands were censused every year.

Year	No. of pairs	No. of marshes with LFCR
1980	203	11
1981	173	15
1982	221	18
1983	249	18
1984	277	19
1985	142	14
1987	178	---

Figure 8.2. Importance of selected southern California wetlands to the region's population of light-footed clapper rails. Data are percent of total birds censused, by year.

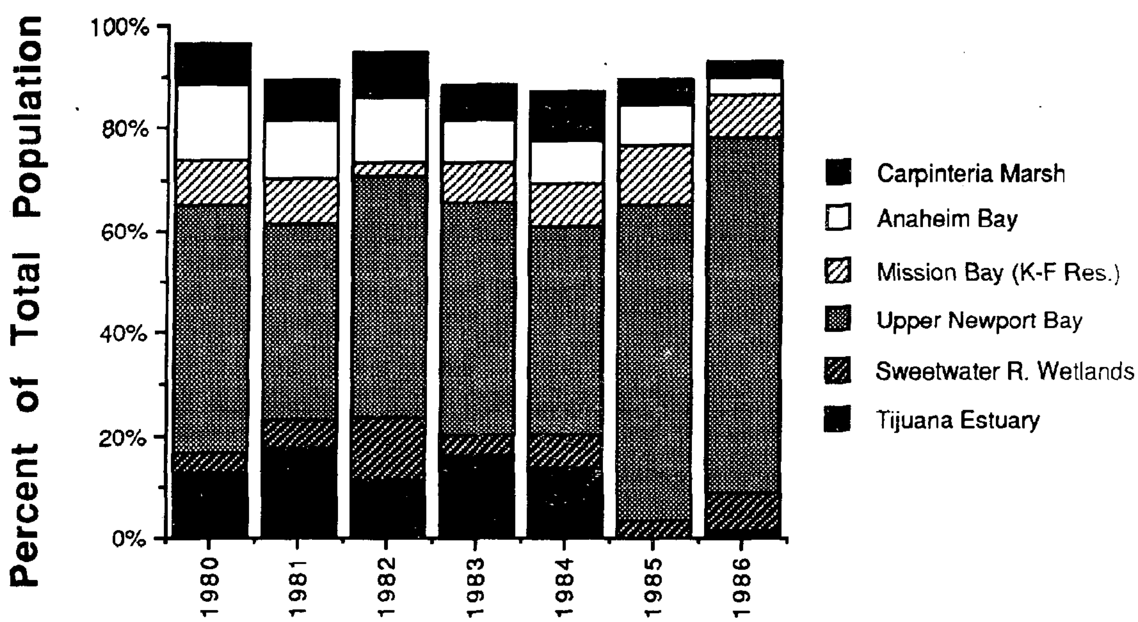
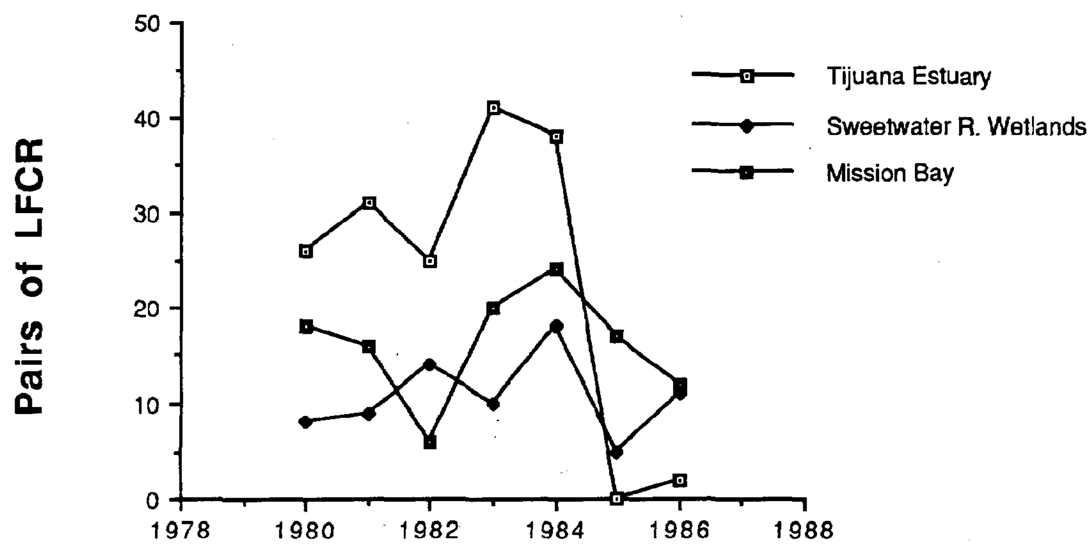


Figure 8.3. Dynamics of the light-footed clapper rail populations at three wetlands in southern California. The population at Tijuana Estuary crashed while the ocean inlet was closed to tidal flow (April to December 1984).



California least terns are monitored for type and extent of habitat use (when present, April-Nov.) in both constructed and reference channel habitats. Feeding rates are recorded, and type and quantity of food use are estimated. The California Dept. of Fish and Game should be consulted to advise on censuses and to exchange information on the abundance of these birds elsewhere.

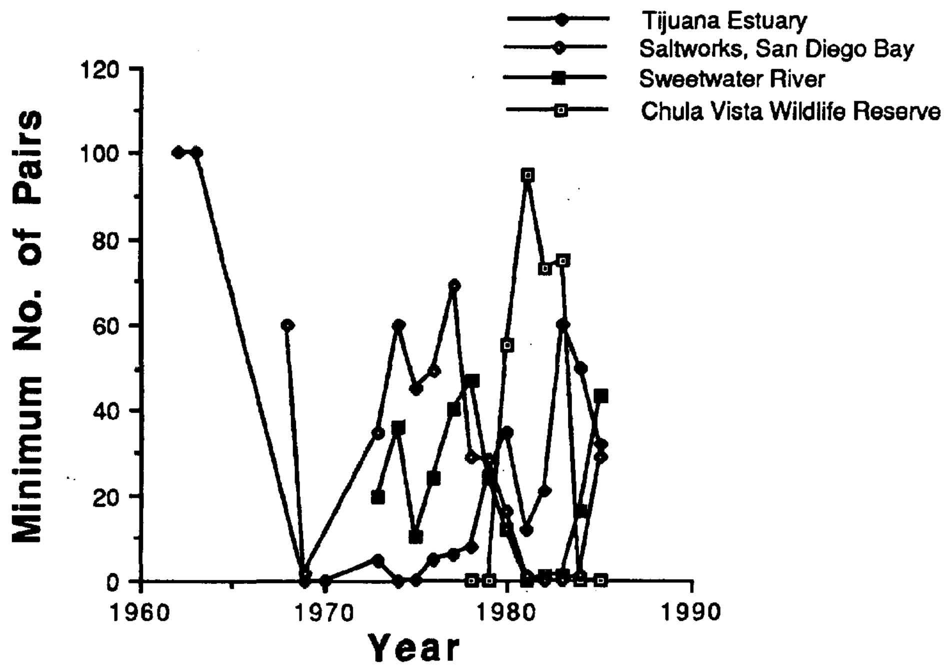
The recovery plan for this endangered species (CLT Recovery Team 1980, p. 13) states that "at least 1,200 pairs distributed among colonies in at least 20 coastal wetland ecosystems throughout their 1977 breeding range" would be required to restore and maintain the breeding population. Historical data (Table 8.7) show that the state-wide population fluctuates, with no clear trend for an increasing density.

Table 8.7. Historical data for California least tern breeding population (for state of California), as summarized in the US FWS Biological Opinion (Steucke 1988).

Census	Pairs
Early 1970's	623-763
1982	1015-1245
1983	1196-1321
1984	887-997
1985	954-1084
1986	936-1016

California least tern
(Sterna albifrons browni)

Figure 8.4. Historical data for California least terns at selected sites in San Diego County (E. Copper, ornithologist and least tern authority, unpub. data).



Belding's Savannah sparrows are censused during their spring territorial nesting period. Territories are defended by singing males, which are best censused between 0600 and 0900 h (White 1986). Observers carry binoculars, walk transects through the entire marsh, and count singing males separately from non-singing birds (which could be either males or females). The counts of singing males estimate the number of breeding pairs. To carry out detailed studies of habitat use, territory sizes should be estimated and mapped according to the method of White (1986).

There are several sources of data comparing populations of Belding's Savannah sparrow. A recent survey (Table 8.8) included 4 wetlands. Zembal et al. (in press) provided data for 31 coastal wetlands from San Diego County to Santa Barbara County, summarizing censuses from 1973, 1977, and 1986. White (1986) gives additional data for Los Peñasquitos Lagoon and Tijuana Estuary.

Table 8.8. Comparative data for the Belding's Savannah sparrow in southern California (A. White, PERL, unpub. data). All censuses were in March 1989 and used the same sampling methods. Numbers are for singing males, which indicate breeding pairs.

	Estimated pairs
Tijuana Estuary	299-320
Sweetwater River Marsh Complex	160-183
Los Peñasquitos Lagoon	56
Ballona Wetland	31

9. Reptiles and amphibians

Objectives. Censuses are made to determine the species present in various habitats. The functions performed by the herpetofauna are not well studied. Species distributions and abundances are also poorly known.

Methods. Pitfall traps were used by Hollis et al. (1988) to characterize reptiles and amphibians at Santa Margarita Estuary. These traps were "5-gallon buckets sunk in the ground with an elevated wood cover. Natural debris was used to construct drift fences to direct the animals toward the traps." (ibid., p. 45). No indication of the numbers of traps or amount of searching was provided. Only a few species were were found (Table 9.1).

Similar methods were employed by R. Espinoza (SDSU, unpub. data) in 1989, with sampling in ten sites ranging from wet to upland areas (Table 9.1). His traps were 5-gal. plastic buckets with holes drilled in the bottom to insure drainage in case of rain. Buckets held sand or shredded newspaper to provide cover for trapped animals and to reduce fighting and predation among captives. Each bucket had an oversized masonite lid, which was propped open during times when traps could be checked. The pitfall traps were placed 14.4 m apart, with 15 buckets for each sampling site. Drift fences were used in conjunction with the pitfall traps, to help direct animals into the traps. Espinoza used aluminum flashing, supported with wooden stakes, and added sand at the base to act as a horizontal barrier. These "fences" were 36 cm high and constructed in a widened "H" shape, with parallel legs of 21 m and a 30.5-m cross link. Five traps were buried along each 21-m leg (the middle trap also encountered the cross link), and 5 along the cross-linking fence. Fencing passed over the middle of each bucket, so that animals moving along either side of the fence would encounter the trap. Espinoza also searched sites, looking under vegetation, debris, rocks and other litter. Animals found were measured from snout to vent, and the condition of their tails noted. Direct observations provide underestimates of nocturnal fauna, so are best used in conjunction with pitfall traps.

Table 9.1. Reference data for herpetofauna at Tijuana Estuary (T) and Santa Margarita Estuary (SM), including willow woodland, coastal strand, maritime scrub and grassy habitats.

Order	Family	Common name	Species	SM	T
Anura - Toads, frogs	Hylidae	Pacific treefrog	*Hyla regilla*	x	x
	Ranidae	Bullfrog	*Rana catesbiana*	x	x
	Pipidae	African clawed frog (exotic)	*Xenopus laevis*		x
	Bufonidae	California toad	*Bufo boreas halophilus*		x
Squamata - Lizards, snakes	Iguanidae	Great Basin fence lizard	*S. occidentalis biseriatus*	x	x
		Side-blotched lizard	*Uta stansburiana*	x	x
		San Diego coast lizard	*Phrynosoma coronatum blainvillei*		x
	Scincidae	Coronado (Island) skink	*Eumeces skiltonianus interparietalis*		x
	Anguidae	San Diego alligator lizard	*Gerrhonotus multicarinatus webbi*	x	x
		Silvery legless lizard	*Anniella pulchra pulchra*		x
	Colubridae	San Diego gopher snake	*Pitouphis melanoleucus annectens*	x	x
		California king snake	*Lampropeltis getulus californiae*	x	x
		Hammond two-striped garter snake	*Thamnophis hammondi hammondi*		x
	Viperidae	Southern Pacific rattlesnake	*Crotalus viridus helleri*	x	
Testudenes Turtles	Emydidae	Southwestern pond turtle	*Clemmys marmorata pallida*	x	

10. Mammals

Objective. Our knowledge of the role of mammals in coastal wetlands is rudimentary. Part of their role in the food chain is feeding on plants and being preyed upon by raptors. Some seed dispersal is carried out by mammals. Their burrowing activities disturb the soil surface and open space for seedling establishment (Cox and Zedler 1986). In constructed and restored wetlands, herbivores can decimate transplants, especially if the soil is not very wet or tidal inundation is lacking. Ground squirrels and rabbits are probably the main grazers on transplants, and a pre-planting survey of the project site would help determine if fencing or herbivore exclusion cages (Zedler 1984) will be necessary. Additional studies are needed to quantify mammal populations in coastal wetlands and to determine their unique roles in the ecosystem.

Methods. Small mammals are trapped using Sherman traps. A convenient way to express the results is as new captures per 100 trap nights. Mark-recapture methods provide more accurate estimates of population size than single trappings.

At Tijuana Estuary, Taylor and Tiszler (1989, unpub.) sampled small mammals in seven habitat types, of which four were in the salt marsh (*Salicornia*, *Salicornia* / salt panne, salt panne, and salt grass), and the remainder in peripheral habitats (border, upland, upland transition). Each was sampled monthly with 2 trapping grids. A grid consisted of 10 Sherman traps located at points 10 m apart in a 2x5-point grid. Trapping was done from November through May, for a total of 1440 traps. Although animals were marked, the low numbers of recaptures precluded density estimates for any of the species.

At Santa Margarita River Estuary (Hollis et al. 1988), 30 trap lines were used in eight habitat types (dense *Salicornia*, *Salicornia* / *Distichlis*, *Salicornia* / salt panne, Upland/*Salicornia*, coastal strand, maritime scrub, cattails, and willow woodland. Each of the lines included 10 traps, which were maintained for four consecutive evenings every 3-4 months.

Reference data. At Tijuana Estuary, there were 114 captures and 8 recaptures during the 7-month study (Table 10.1). Only three rodent species occurred in the marsh. These were the western harvest mouse *(Reithrodontomys megalotis)*, the deer mouse *(Peromyscus maniculatus)*, and the house mouse (*Mus musculus*). Five additional species were trapped in Goat Canyon, an upland area. The species list was extended with evidence from sightings, scat, skeletons, and pellets.

At Santa Margarita River Estuary, Hollis et al. (1988) and previous surveys recorded 22 species. Hollis et al. (ibid.) found the highest number of species per habitat (9) in the maritime scrub habitat, while the largest number trapped was in the willow woodland (Table 10.2). There were seasonal changes in numbers trapped. Low numbers were associated with inlet closure and marsh flooding; in 1987, high densities in Feb. and May were followed by low densities in August.

Table 10.1. Mammals known from Santa Margarita River Estuary (SM; from Hollis et al. 1988) and Tijuana Estuary (TE; Taylor, SDSU, unpub. data).

Order	Family	Common name	Scientific name	SM	TE
Marsupiala	Didelphiidae	Opossum	*Didelphis marsupialis*	x	x
Insectivora	Soricidae	Ornate shrew	*Sorex ornatus*	x	
Lagomorpha	Leporidae	Blacktail jackrabbit	*Lepus californicus*	x	x
	Audubon cottontail		*Sylvilagus auduboni*	x	x
Rodentia	Sciuridae	Calif. ground squirrel	*Spermophilus beecheyi*	x	x
	Geomyidae	Botta pocket gopher	*Thomomys bottae*	x	
	Heteromyidae	San Diego pocket mouse	*Perognathus fallax*	x	x
		Agile kangaroo rat	*Dipodomys agilis*		x
	Cricetidae	Western harvest mouse	*Reithrodontomys megalotis*	x	x
		Deer mouse	*Peromyscus maniculatus*	x	x
		Cactus mouse	*Peromyscus eremicus*		x
		California mouse	*Peromyscus californicus*	x	
		Brush mouse	*P. boylii*	x	x
		Desert wood rat	*Neotoma lepida*	x	
		Dusky-footed wood rat	*N. fuscipes*	x	
		California meadow mouse	*Microtus californicus*	x	x
	Muridae	Norway rat	*Rattus norvegicus*	x	
		House mouse	*Mus musculus*	x	x
Carnivora	Procyonidae	Raccoon	*Procyon lotor*	x	
	Mustilidae	Long-tailed weasel	*Mustela frenata*	x	x
		Striped skunk	*Mephitis mephitis*		x
	Canidae	Coyote	*Canis latrans*		x
Artiodactyla	Cervidae	Mule deer	*Odocoileus hemionus*	x	

Table 10.2. Results of small mammal trapping at Santa Margarita River Estuary (summarized from Hollis et al. 1988).

Uncommon species (averaged ≤5 new captures per 100 trap nights in any habitat):	Ornate shrew Botta pocket gopher Pocket gopher California mouse	Brush mouse Desert wood rat Dusky-footed wood rat Audubon's cottontail
Common species:	Western harvest mouse (WHM) Deer mouse (DM) California meadow mouse (CMM) House mouse (HM)	

Mean new captures per 100 trap nights (by habitat type) for the 4 common species:

Habitat type	WHM	DM	CMM	HM
Dense *Salicornia*	13.9	4.6	11.9	7.8
Salicornia/Distichlis	9.7	4.4	3.5	2.5
Salicornia/saltpan	1.6	12.6	0.3	0.2
Upland/*Salicornia*	9.5	9.4	1.6	4.1
Coastal strand	3.3	19.5	0	5.0
Maritime scrub	3.8	14.6	3.8	0.6
Cattails	4.3	18.6	7.7	9.3
Willow woodland	2.3	19.7	4.0	6.2

V. Recommendations for minimum monitoring

We recognize that most assessment and monitoring programs will be constrained by funding and by the availability of personnel who are qualified to sample such things as nitrogen fixation. Local resource agencies have requested that we recommend minimal requirements for use in setting permit conditions for wetland restoration or construction projects. Since the main purpose of monitoring is to characterize the structure and functioning of the wetland, the sampling program should be able to withstand the review of field ecologists. Thus, the program should identify the habitats being characterized; it should have replicate sampling stations within each habitat; and it should provide data that document ecologically meaningful changes when they occur. General analyses of the data should indicate that the sampling program is encountering the bulk of the species present, and that variances among replicate sampling stations are not excessively high. The recent text on "Ecological Methodology" (Krebs 1989) provides further discussion of these issues.

Monitoring programs can be expanded or reduced in different ways, by varying the number of attributes examined, the frequency of examination, and the number of sampling stations. Additional cuts or additions could include the detail of examination within sampling stations (e.g., depths at which soil salinity is measured) and back at the laboratory (e.g., determination of invertebrates to family or to species; chemical analysis of pooled or individual soil samples from each sampling station).

Priority attributes. The attributes can be prioritized based on what we need to know and how much information is provided by the data (priority 1 = most needed; 2 = desirable; 3 = worthwhile). It should be noted that our priority designations are tentative; as we come to understand more about wetland ecosystem functioning, we are more able to select indicators of function. In the future, it may become clear that some priority 3 variables are essential measurements, or *vice versa*.

Sampling frequency. As discussed earlier, ecosystem attributes differ in their temporal variability. Birds come and go daily, and large changes occur with the fall and winter migratory periods. In contrast, plant invasions or local extinctions usually become obvious only after a year or two. Thus, we recommend that some attributes be measured as often as weekly, others seasonally or annually, and some only after major events are noted.

Not all wetlands will have the same temporal variability, so it is difficult to suggest a single program that can fit all systems. Soil salinities, for example, will show greater extremes and more sudden changes in lagoon wetlands (often nontidal) than in fully tidal marshes. Monitoring programs should be tailored to the needs of the system being monitored, beginning with frequent measurements and reducing sampling if experience suggests that reducing the frequency will not significantly reduce information about the system. Monitors should be prepared to increase sampling frequency in response to events such as floods, wastewater spills, algal blooms, or inlet closure.

Numbers of sampling stations. Field monitoring programs should provide an adequate sample of the area to which results will be generalized. Experienced field ecologists can usually walk through a site and delimit habitat areas that are "relatively homogeneous," but aerial photos are a great aid. Within each habitat area, replicate samples are taken at no fewer than three stations. Initial sampling will provide estimates of heterogeneity (variance of each attribute measured); if initial replicate stations give high variance (e.g., if the standard error exceeds 10% of the mean), additional replicate samples are needed to characterize the attribute adequately. Because the system's variability dictates the number of replicate samples needed, we cannot prescribe the number of sampling stations needed. Our advice is to plan for a large number of replicate stations and cut back if variances are low. Krebs (1989) discusses the numbers of replicate samples needed to provide ecologically meaningful data. Results can be summarized to test for differences between different locations (e.g., restored and natural wetlands) or differences with time (e.g., year-to-year changes).

An alternative approach to replicate sampling within habitat areas is appropriate where gradients of environmental conditions are present. For estuarine channels that range from high salinity at the inlet to low salinity inland, it is more useful to position sampling stations along the gradient and to plot water quality characteristics against distance. Instead of clumping sampling stations within homogeneous sampling areas, one would distribute the stations at intervals proceeding upstream from the ocean inlet. Stations should be closer together where environmental changes are likely to be greatest. Results can be summarized as graphs of each attribute against distance from inlet, looking for spatial trends and evidence of shifts through time.

How long to monitor. From the standpoint of the biota, a twenty-year monitoring period is not unreasonable. It may take even longer for the restored marsh to develop its full potential as habitat for rare species, such as endangered birds. It may take longer for the soil organic matter to increase to natural levels. It may take longer for herbivory problems to become controlled by native predators. Finally, for a region that has highly variable rainfall, it may take 20 years to characterize the mode, or most usual condition of the wetland, should the monitoring period include years of unusual events. Such was the case at Tijuana Estuary, where salt marsh monitoring began in 1979, two major floods occurred in winter 1980, prolonged flooding occurred through April 1983, the tidal inlet closed for 8 months in 1984, and raw sewage inflows became substantial and continuous in about 1986.

Long-term monitoring allows one to distinguish directional changes (e.g., expansion of cordgrass, declines of endangered bird populations) from short-term shifts (e.g., annual variability in shorebird use). Permits for projects in wetlands sometimes require 20 years of monitoring (Phil Williams, Phil Williams Assoc., San Francisco, pers. comm.) and often require up to 10 years of assessment (with annual measures in the earlier years).

Paul Zedler analyzed San Diego rainfall data as a measure of the variability of vernal pool environments (Zedler and Black 1989). For the 138 years of record, the range was 57 cm, which is about twice the average annual rainfall. He calculated that monitoring would need to take place for 7-8 years just to include half that range. Studies of 2-3 years would typically cover only 20-30% of the historic range. Along a similar vein, Westeby (in press) states, "Studies lasting 10-50 years do not provide enough replicate years to reliably detect any but the strongest correlations between variables at one site, nor do they estimate frequencies of rare events very well."

Conclusion. Choosing a sampling program that can provide ecologically meaningful data through a ten-to-twenty-year period is not easy. Dozens of decisions need to be made, and most require careful judgment based on preliminary data from the system in question. The assistance of experienced field workers will be needed to tailor any "generic monitoring program" to the system in question.

Table 4.1. Priorities for wetland attributes to be monitored. Priority 1 = most needed.

Attribute and measures	Priority	Frequency of measurement
Hydrology		
Salinity of water	1	monthly
Salinity of interstitial soil water	1	seasonally (at least Apr. & Sept.)
Water levels at various tidal cycles	2	spring tide cycle, e.g., in Nov.
Tidal flow rates at distances from inlet	3	spring tide cycle, e.g., in Nov.
Topography		
Elevation	1	initially and after storms or floods
Slope of channel banks	3	initially; annually thereafter at permanent cross-sections
Soils		
Texture	2	initially
Organic matter	1	pre-planting; plan for amendments
Toxic substances	3	pre-dredging; costly analysis
Redox potential	2	useful in diagnosing cause of plant mortality
Sulfides and pH	3	redox potential is easier
Nutrient dynamics		
Nitrogen fixation rates	3	seasonal, as research study
Inorganic nitrogen in sediments and pore water	1	initially to plan for amendments; repeat if plant growth is poor
Litter decomposition	3	seasonal, as research study
Nitrogen mineralization rates	3	seasonal, as research study
Foliar nitrogen of dominant plant species	2	annually in September
Algae		
Cover by dominant type	1	monthly with salinity samples
Vascular plants		
Aerial photos of plant cover and habitat types	1	annually
Heights and total stem length of cordgrass	1	annually in September
Cover of vascular plants	1	annually in September
Patch size of rare annual plants	1	annually in spring
Density of annual plants	3	annually in spring
Consumers		
Decomposers and shredders	3	seasonal, as research study
Aquatic insects	2	seasonally
Terrestrial insects,	3	seasonal, as research study
especially pollinators	1	important where annual plants are required; census in spring
and predatory insects	1	important where insect herbivory is obvious; census in warm season (May-Oct.)
Fishes	1	at least in June and September
Benthic invertebrates	1	seasonally
Birds	1	weekly in fall-spring; biweekly in summer
Reptiles and amphibians	3	summer
Mammals	3	(high priority in regions with salt marsh harvest mouse)

Table 4.2. A minimum monitoring program for a wetland with aquatic and marsh habitats.

	Before project	J	F	M	A	M	J	J	A	S	O	N	D
		Annual monitoring schedule											
Hydrology													
Salinity of water		X	X	X	X	X	X	X	X	X	X	X	X
Salinity of interstitial soil water		x			X			x		X			
Topography													
Elevation	X	and after storms or floods											
Soils													
Organic matter	X	repeat 2-3 months after any soil amendments											
Nutrient dynamics													
Inorganic nitrogen in sediments and pore water	X	repeat 2-3 months after any soil amendments											
Algae													
Cover by dominant type		X	X	X	X	X	X	X	X	X	X	X	X
Vascular plants									X				
Aerial photo analysis													
Heights and total stem length of cordgrass										X			
Cover of vascular plants										X			
Patch size of rare annual plants					X								
Consumers													
Fishes		x			X			X		X			
Benthic invertebrates				X			X			X			X
Birds	W	W	W	W	W	B	B	B	B	W	W	W	

X = sample once during month; x = sample to omit if funds are insufficient;
W = sample weekly through month; B = sample biweekly.

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